UNIVERSITY
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Re-establishing dry sclerophyll forest on unused farmland

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Abstract

There have been few systematic experimental studies testing different restoration techniques in varying environments on the same property. The present experimental work was done to provide a basis for re-establishing sclerophyll forest on agricultural land located on the Tasman Peninsula. In determining appropriate habitat restoration goals it was necessary to reconstruct the original vegetation. Thus the first goal was to conduct a vegetation survey of the field site with the aim of identifying and mapping the vascular plant species and community types present. The results from the vegetation survey showed that the remnant vegetation consisted of sedgey *Eucalyptus ovata* woodland, shrubby *E. obliqua* forest and heathy *E. obliqua/E. amygdalina* forest and that the pasture consisted of five floristic communities dominated by exotic grasses. The environmental envelopes of the three native vegetation communities were then compared to those of the exotic vegetation communities. This process allowed the exotic species combinations to be used as analogues of the former distribution of the native vegetation communities.

The level of intervention that was required to promote the successful re-establishment of native vegetation was investigated. The sowing of native seeds resulted in poor germination of all native species, with the exception of *Acacia verticillata* and *A. melanoxylon*, with germination the greatest in unfenced areas where weed colonisation of gaps was slowest. The reintroduction of *Eucalyptus* and *Allocasuarina* seedlings into three different exotic weed communities demonstrated that soil properties had a greater influence on seedling survival and growth than competition from exotic weeds and that species did best on sites in which they would have naturally occurred.

In the field of restoration ecology there is little research on whether the simultaneous reintroduction of tree, shrub and groundcover species could accelerate forest restoration on agricultural land compared to plantings of single species. The results obtained from the field trial that examined the neighbour interactions of *E. obliqua*, *Allocasuarina*

littoralis and *Poa labillardierei* showed that the presence of neighbours could promote greater seedling growth than that of individuals planted without neighbours. For *A. littoralis* and *E. obliqua* the greatest growth occurred when its neighbours were four *A. littoralis* seedlings, whilst the greatest growth by *P. labillardierei* occurred when its neighbours were a mixture of itself and *E. obliqua*. These results suggest that intra- and interspecific interactions can have a positive effect on the growth of the three species.

The field trials at the study site indicate that competition from weeds and lack of native seed storage limit the ability of native vegetation to re-establish on the abandoned pasture. Poor establishment from the sowing of native seeds onto gaps created in the grass sward illustrated the need to re-introduce seedlings to promote the restoration of native species in the exotic weed communities. The seedlings of different dominant tree species varied in their survival and growth in different exotic weed communities, indicating the importance of adjusting restoration processes to variations in microhabitat. Restoration may also be accelerated by planting mixtures of tree, shrub and groundcover species.

Acknowledgements

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Figures and Tables

Chapter One

Introduction

The restoration of native vegetation communities, ecosystem health and productivity on degraded agricultural land has generally relied on abandonment of land and subsequent forest succession (Parrotta *et al.* 1997). However, the ability of degraded lands to return to a natural or historical state is usually limited by abiotic and biotic barriers to restoration (Whisenant 1999; Cramer *et al*. 2008; Gomez-Aparicio 2009). Impoverished soil seed banks and limited seed dispersal can be biotic barriers to the recovery of degraded lands (Zimmerman *et al*. 2000; Standish *et al*. 2007) whereas abiotic barriers include soil properties and soil nutrient levels which differ dramatically from their historical state (Standish *et al*. 2006; Walker and Reddell 2007).

The recolonisation of sclerophyll vegetation on cleared land after cultivation is rare and where it does occur it is generally restricted to areas adjacent to undisturbed native forest (Withers 1978; Grubb and Hopkins 1986; McDonald 1996; Yates and Hobbs 1997a). This slow natural re-establishment of native vegetation, along with the widespread clearance of *Eucalyptus* dominated forest and woodland in temperate Australia, has led to a growing interest in the development of restoration strategies.

The diversity and isolation of the many groups of people involved in restoration work in Australia has meant that much activity has been undertaken without an appreciation of the ecological processes involved, emphasising a clear need to better integrate science and practice (McDonald and Williams 2009). Determination of the biotic and abiotic barriers preventing native establishment is considered a key factor in increasing restoration success with most studies identifying dispersal, weed competition and poor soil conditions as important barriers (Saunders and Hobbs 1995; Zimmerman *et al.* 2000). Hooper *et al*. (2005) found that the interaction of limited seed dispersal and competition with invasive exotic species either slowed the establishment of native

species on abandoned farmland or allowed the establishment of only a subset of species with recruitment and establishment traits that render them competitive with exotic species.

The removal of competition for resources from exotic vegetation is considered one of the most effective techniques to simultaneously overcome the abiotic and biotic barriers to native vegetation restoration using such treatments as fire, herbicides, grazing or mechanical removal (Whisenart 2002; Gomez-Aparicio 2009). Other restoration strategies usually centre on accelerating changes in species composition through the re-introduction of native plants via direct seeding or tubestock planting to overcome the lack of native seed dispersal in the degraded system (Pyke and Archer 1991; Walker *et al*. 2007).

In old fields of the West Australian wheatbelt, Standish *et al*. 2007 found that in most cases, direct seeding and control of non-natives were the minimum requirements for restoration of species-rich eucalypt woodlands. Whereas the restoration of eucalypts and other woody species that recruit after natural disturbance events, such as fire, may require the application of techniques that mimic that disturbance. Grazing by livestock is also thought to be a primary factor currently limiting eucalypt recruitment in agricultural landscapes (Reid and Landsberg 2000).

The wide range of soil and site conditions of abandoned farmland, as well as land use histories, means that strategies developed in one particular region would not necessarily work in other regions, emphasising the importance of understanding the basic biology of the local ecosystem to design effective restoration strategies (Holl *et al.* 2000).

Despite the above problems, it has been suggested that the integration of information from different combinations of experimental and quantitative tools can be used to predict restoration success at a general level (MacMahon and Holl 2001). Thus, comprehensive scientific investigations at the local scale may then be utilised to provide

a framework on which to build generalised models of ecosystem development on abandoned farmland (Hobbs and Cramer 2007).

Thesis aims and structure

This thesis aims to provide a basis for re-establishing sclerophyll forest on abandoned farmland by investigating:

- the parameters for restoration in a particular area; and
- whether general restoration issues may be addressed from the experimental findings.

The current study was undertaken on a partly cleared property at Saltwater River on the Tasman Peninsula, Tasmania and was devised to help restore the present agricultural landscape to a more "natural" landscape.

Chapter 2 presents the findings from a vegetation survey of the site which had the aim of identifying and mapping the vascular plant species and community types. This information was then used to reconstruct the native vegetation communities of the study area before agricultural disturbance, thus providing a template for restoration of the site.

The aim of Chapter 3 was to examine the factors which potentially influence seedling recruitment of native forest vegetation in the improved pasture zones of the study site. A field experiment was conducted to examine seedling emergence of four tree and shrub species, *Eucalyptus obliqua, Allocasuarina littoralis, Acacia melanoxylon* and *A. verticillata*, typical of the remnant forest bordering the pasture.

Chapter 4 investigates whether the variation in topography and drainage conditions across the site is likely to affect the survival and growth of *Eucalyptus* seedlings. The questions that were addressed are 1) Does site selection based on a species natural distribution optimise seedling survival and growth? 2) Which weed control treatment is

most effective for promoting seedling growth? 3) Is there a significant difference between sites in the effectiveness of weed control treatments?

Chapter 5 investigates whether the recovery of *E. obliqua* open-forest on abandoned farmland can be accelerated by the simultaneous re-introduction of tree, shrub and groundcover species. Information on the interactions of *E. obliqua* with neighbouring plants mainly comes from silvicultural studies and thus little is known about interactions between *E. obliqua* seedlings and native shrub and groundcover species.

Chapters 2 through to 5 have been written in the format of journal articles, and therefore have individual Introduction and Discussion sections. Overall conclusions and practical implications of the experimental work are provided in Chapter 6.

Chapter Two
Reconstruction of native vegetation communities

Introduction

A key factor in determining the success of ecological restoration is the overall correspondence of the restored community to the undisturbed remnant community (Cairns 1993). To determine the vegetation communities to be restored on sites that have been modified by agricultural disturbance it is important to have an understanding of the historical background (Duggan 1993; Fry and Main 1993; Saunders and Hobbs 1995). Another important goal of restoration is restoring ecosystem functions and preserving the ecological and genetic balances of the adjacent native communities (Handel *et al.* 1994; Lesica and Allendorf 1999). The living model of remnant vegetation as a reference point can provide ecological details, such as the interaction among habitat types that is necessary for survival of component species (Handel *et al.* 1994). Thus, using the remnant and neighbouring native vegetation as a template for ecological restoration provides critical information for determining restoration goals.

A survey of the restoration site is also recommended to ascertain the range and type of plant communities present along with the environmental characteristics of the communities. The soil properties, aspect, drainage and altitude of disturbed habitats can be used as indicators of previous native communities (Duncan and Brown 1985; Anderson 1989). In one example, if "soils are naturally wet, base-rich boulder clays, appropriate habitats would be marshes, damp grassland or woodlands. If the soils are sandy or acidic, heathland creation may be more appropriate" (Anderson 1989, p 236).

The current study, on a property at Saltwater River on the Tasman Peninsula, Tasmania was devised to help restore the present agricultural landscape to a "natural" landscape. The property contains a mixture of *Eucalyptus* forest, wetland, scrub and pasture. Mean

annual rainfall is c. 780 mm, peaking in August (80mm) with the minimum in February (50 mm). To obtain a reference point for the restoration of native vegetation at the site, a vegetation survey was conducted with the aim of identifying and mapping the vascular plant species and community types. The environmental characteristics of the remnant vegetation communities were then compared to those of the exotic vegetation communities. This process was used to reconstruct a map of the native vegetation communities of the study area before agricultural disturbance, thus providing a template for restoration of the site. The reconstruction of the native vegetation communities was in turn used to identify which native vegetation species should be the focus of the scientific investigations undertaken to identify the restoration techniques required to overcome barriers to natural restoration at the site (refer to Chapters 3, 4 and 5).

Materials and methods

Vegetation survey

The vegetation survey was undertaken from November 1999 to January 2000 and consisted of stratified sampling across ten vegetation mapping units that were identified from aerial photographs and a preliminary survey. Using a table of random numbers five sampling locations were placed in each unit. At each sampling site a quadrat was laid out. As the communities varied in structure, the size of the area sampled was adjusted to account for the variation in the scale and fragmentation of the communities. Determination of appropriate quadrat size took into consideration the morphology of the vegetation species and the homogeneity of the vegetation in each community (Mueller-Dombois and Ellenberg 1974; Chapman 1976; Critchley and Poulton 1998). Based on these considerations, three quadrat sizes were used, however as advised by Critchley and Poulton (1998) a constant size was used within a particular vegetation type. The quadrats used to sample the four forest communities were 10 m × 10 m, while 5 m × 5 m quadrats were used to sample the four grassland and wetland scrub communities and 2 m × 2 m quadrats sampled the *Carex appressa* community.

All observable and identifiable vascular plant taxa within the quadrat were noted. Where possible, vascular plant taxa were identified in the field using field guides and botanical keys (Curtis 1975; Lane *et al.* 1999) with the assistance of Prof Jamie Kirkpatrick (University of Tasmania). Samples of any unidentified taxa were collected for further examination and identified with the assistance of staff at the Tasmanian Herbarium. The cross-sectional area of trees at 1.3 m (basal area) was measured along with the height and cover of the canopy. The canopy cover, shrub stratum cover and leaf litter were estimated using the following classes: <1%, 1-5%, 6-25%, 26-50%, 51-75%, and 76-100%. Along with floristic information, slope and aspect were recorded for each quadrat. Field observations were made of areas that were not represented by quadrats to help in drawing the vegetation boundaries.

Data analysis

A sorted table of the species by site data was produced using the computer program TWINSPAN (Hill 1979). The table grouped quadrats with similar species composition, using presence/absence data, and was used to select the floristic communities (Kirkpatrick 1999; O'Shea and Kirkpatrick 2000).

An ordination (global non-metric multi-dimensional scaling using the default options in DECODA (Minchin 1991)) of the floristic data was also carried out to identify the groups of quadrats that had similar floristic composition. The data used in the classification and ordination were the plant taxa that were present in at least two quadrats.

To help reconstruct the vegetation communities before agricultural disturbance the physical attributes which were related to the distribution of the remnant communities were measured. An one-way ANOVA was then carried out to determine which communities had statistically similar physical attributes.

Results

The flora

One hundred and sixty vascular plant taxa were identified from the study area (Appendix 2.1). One hundred and twenty-two (76 %) of these taxa were natives, four (3 %) of which are endemic to Tasmania, while thirty-eight (23 %) were introduced species. The tree stratum was dominated by taxa from the Myrtaceae family, with five *Eucalyptus* species recorded. Asteraceae, Fabaceae, Epacridaceae and Mimosaceae dominated the shrub stratum. The native ground layer species were largely represented by the Asteraceae, Cyperaceae, and Juncaceae families. Of the introduced flora, the dominants consisted of species from the Poaceae (grasses) and Asteraceae (daisy) families.

The plant communities

The sorted table (Table 2.1) groups the quadrats (numbered across the top of the sorted table) that were identified by TWINSPAN as having similar floristic composition. These groups of quadrats are the floristic communities that were identified through the presence or absence of the groups of species (lettered) that run down the sorted table. Eight floristic communities were observable from the sorted table.

Communities 1, 2 and 3 were *Eucalyptus* forest communities defined by the presence of species in groups A to D, while communities 4 to 8 were grassland and sedgeland defined by the presence of species in groups E to K. In Figure 2.1 the fifty quadrats classified by community have their ordination values plotted to indicate floristic similarities. The communities that had a higher diversity, such as the three forest communities, had a larger spread of plots while the less diverse grassland communities had their quadrats grouped closely together. Communities 1, 2 and 3 have a similar floristic composition, which is different to the floristic composition of communities 4, 5, 6, 7 and 8.

Table 2.1: Sorted table for species that occurred in two or more quadrats at the Saltwater River study site.

		1144444	133333334	3	3	1	44445	112211	22231	122222
		2541235	383672590	4541231	67908	67890	790868	67904	141235	
	Community	1	2	3	4	5	6	7	8	
	Ranunculus spp.	---1--1	---------	-------	-----	-----	------	-----	------	
	Zieria arborescens	---11--	---------	-------	-----	-----	------	-----	------	
	Blechnum nudum	--111-1	---------	-------	-----	-----	------	-----	------	
A	*Drymophila cyanocarpa*	1---1-1	-1-------	-------	-----	-----	------	-----	------	
	Stipa spp.	--1----	----1----	-------	-----	-----	------	-----	------	
	Senecio linearifolius	---1-1-	----1---1	-------	-----	-----	------	-----	------	
	Poranthera microphylla	--111--	----1--11	-------	-----	-----	------	-----	------	
	Diplarrena moraea	-1-----	-1-11----	-------	-----	-----	------	-----	------	
	Oxylobium arborescens	-------	11-------	-------	-----	-----	------	-----	------	
	Wahlenbergia spp.	-------	--11-----	-------	-----	-----	------	-----	------	
	Monotoca glauca	-------	-1-1-----	-------	-----	-----	------	-----	------	
B	*Banksia marginata*	-------	---11----	-------	-----	-----	------	-----	------	
	Cassinia aculeata	-------	--111---1	-------	-----	-----	------	-----	------	
	Helichrysum scorpioides	-------	----1---1	-------	-----	-----	------	-----	------	
	Dianella tasmanica	-------	-1--1-1-1	-------	-----	-----	------	-----	------	
	Richea procera	-------	11--1----	-------	---1-	-----	------	-----	------	
	Billardiera longiflora	-----1-	-1-------	1-1----	-----	-----	------	-----	------	
	Clematis aristata	-----1-	-------1-	------1	-----	-----	------	-----	------	
	Eucalyptus ovata	1-1----	-----1---	-----11	-----	-----	------	-----	------	
	Acacia verticillata	1111111	-----11-1	--1----	-----	-----	------	-----	------	
	Olearia lirata	--11111	-1--11111	1-11--1	-----	-----	------	-----	------	
C	*Lagenifera stipitata*	-111---	-1-111111	111-11-	-----	-----	------	-----	------	
	Acacia dealbata	--11---	---1---11	11-----	-----	-----	------	-----	------	
	Eucalyptus obliqua	---1111	-1--11111	11111-1	-----	-----	------	-----	------	
	Viola hederacea	--1-11-	--1-11111	1---11-	-----	-----	------	-----	------	
	Luzula spp.	--1---1	-1-111111	---1111	-----	-----	------	-----	------	
	Corybas spp.	----11-	-----1-11	111111-	-----	-----	------	-----	------	
	Bursaria spinosa	------1	---1111--	--1--1-	-----	-----	------	-----	------	
	Exocarpos cupressiformis	----1--	--1-1111-	11--1-1	-----	-----	------	-----	------	
	Brachyscome spathulata	-------	----1----	1------	-----	-----	------	-----	------	
	Stylidium graminifolium	-------	11-11-1-1	-1-----	-----	-----	-1----	-----	------	
	Astroloma humifusum	-------	1-1---1--	1------	-----	-----	------	-----	------	
	Daviesia ulicifolia	-------	---11-1-1	11--1--	-----	-----	------	-----	------	
	Galium spp.	-------	-----1--1	-----1-	-----	-----	------	-----	------	
	Poa labillardierei	-------	--111-11-	--1----	-----	-----	------	-----	------	
	Lomandra longifolia	-------	--1-1-111	11111-1	-----	-----	------	-----	------	
	Microsorum diversifolium	-------	--1--11--	------1	-----	-----	------	-----	------	
	Bedfordia salicina	-------	--1------	111111-	-----	-----	------	-----	------	
D	*Craspedia glauca*	-------	------1--	--111--	-----	-----	------	-----	------	
	Epacris impressa	-------	11--1----	11-----	-----	-----	------	-----	------	
	Pultenaea juniperina	-------	---1-----	11-1---	-----	-----	------	-----	------	
	Eucalyptus amygdalina	-------	-1--1----	-1----1	-----	-----	------	-----	------	
	Pimelea drupacea	-------	-------1-	----111	-----	-----	------	-----	------	
	Pterostylis spp.	-------	-1-------	----1--	-----	-----	------	-----	------	
	Eucalyptus viminalis	-------	--1-----1	1-1--11	-----	-----	------	----1	------	
	Chiloglottis gunnii	-------	-----1---	-1-111-	---1-	-----	------	-----	------	
	Rubus fruticosus	-------	--1------	-------	-----	-----	------	-----	-1----	
	Picris spp.	-------	1--1-----	-------	-----	-----	------	----1	------	
	Gonocarpus tetragynus	-------	---------	1------	---1-	-----	------	-----	------	
	Acacia melanoxylon	-------	---------	-----1-	-----	-----	------	----1	------	
	Chiloglottis spp.	-111-11	11--111-1	-11111-	---1-	-----	------	-----	------	
E	*Gonocarpus teucrioides*	-1-----	-1--111--	---11-1	---1-	-----	------	-----	------	
	Ehrharta stipoides	1111-11	1-1111111	------1	11-1-	-----	------	-----	------	
	Pteridium esculentum	11-1111	111111111	1111111	11111	1----	------	--1-1	1--1--	
	Oxalis perennans	--1----	--111-111	1----1-	----1	1---1	------	-----	------	
	Acaena novae-zelandiae	-11----	---11----	---11--	-----	1----	11-11-	-----	1-----	
	Senecio spp.	-111111	--1111-11	--1--11	----1	11111	1111-1	-----	1-----	
F	*Geranium* spp.	-11----	--11--111	--11111	-----	1----	11-11-	-----	------	
	Hydrocotyle hirta	--11111	-----11--	------1	-----	1-111	1-1-11	-----	------	
	Gnaphalium spp.	--1----	--1---1-1	-11----	-----	-----	-111-1	-----	------	
	Poa spp.	-------	--111-11-	1-111--	-----	11111	-11---	-----	------	
	Danthonia spp.	-------	---------	11---1-	-----	11111	------	-----	------	

	Agrostis spp.	-------	---------	1--111-	-----	-----	-1---1	-----	------
	Polystichum proliferum	-------	---------	---1-1-	-----	-----	------	-----	------
G	*Pelargonium australe*	-------	---------	---111-	-----	-----	------	-----	------
	Drosera peltata	1------	---------	11-111-	-----	-----	------	-----	------
	Coprosma quadrifida	--1-1-1	---------	---1111	-----	-----	------	-----	1-----
	Isolepis spp.	-------	---------	-------	-----	-----	--1-11	-----	------
	Juncus bufonius	-------	---------	-------	-----	-----	--1-11	-----	------
	Montia australasica	---1---	---------	-------	-----	-----	----1-	-----	------
H	*Lobelia* spp.	-------	---------	-------	-----	-----	----11	----1	------
	Centaurium erythraea	-------	--1------	-------	-----	-----	1-----	--1--	------
	Bromus spp.	-------	---------	-------	-----	----1	111111	-----	------
	Aira spp.	-------	--1-111-1	-------	-111-	----1	------	-----	------
	Hypochoeris radicata	11-----	----1----	-------	1111-	-----	------	--111	------
	Leontodon taraxacoides	-------	------1--	-------	1111-	-----	----11	-----	-1----
	Cerastium spp.	1-1-1--	--1-11111	-------	-11--	1111-	111--1	-----	------
I	*Acetosella vulgaris*	-------	--11-----	-------	11111	11-11	1111-1	1111-	------
	Anthoxanthum odoratum	-------	---11---1	-------	1111-	11111	11111-	11111	11--11
	Holcus lanatus	-------	-11111--1	-------	11111	11111	111111	11111	11-111
	Trifolium repens	-------	---------	-------	1----	11111	111111	11111	------
	Trifolium spp.	-------	---------	-------	1----	11111	111111	11111	------
	Cirsium vulgare	-------	--1------	-------	-----	1111-	11-1-1	-----	------
	Anagallis arvensis	-------	--1------	-------	-----	11--1	11---1	---1-	------
J	*Juncus* spp.	11-----	1--------	-------	-----	-----	111111	11111	111111
	Cotula reptans	-1-----	---------	-------	-----	-----	-1-11-	---11	-1----
	Juncus pallidus	-------	---------	-------	-----	-----	------	1111-	------
	Melaleuca squarrosa	11-----	1--------	-------	-----	-----	------	----1	----1-
	Lepidosperma elatius	-1-----	---------	-------	-----	-----	------	-----	----1-
	Goodenia ovata	-11-111	--1------	1------	-----	-----	------	-----	---1--
K	*Carex appressa*	--11111	---------	-------	-----	-----	-----1	-----	1-1111
	Rubus spp.	-1-----	--1------	-------	-----	-----	------	-----	11----
	Gahnia grandis	---1---	---------	-------	-----	-----	------	-----	1-----
	Hydrocotyle pterocarpa	-11---1	---------	------1	-----	-----	-----1	----1	-111-1
	Azolla filiculoides	-------	---1-----	-------	-----	-----	------	-----	111111
	Triglochin procera	-------	---------	-------	-----	-----	------	-----	-1---1

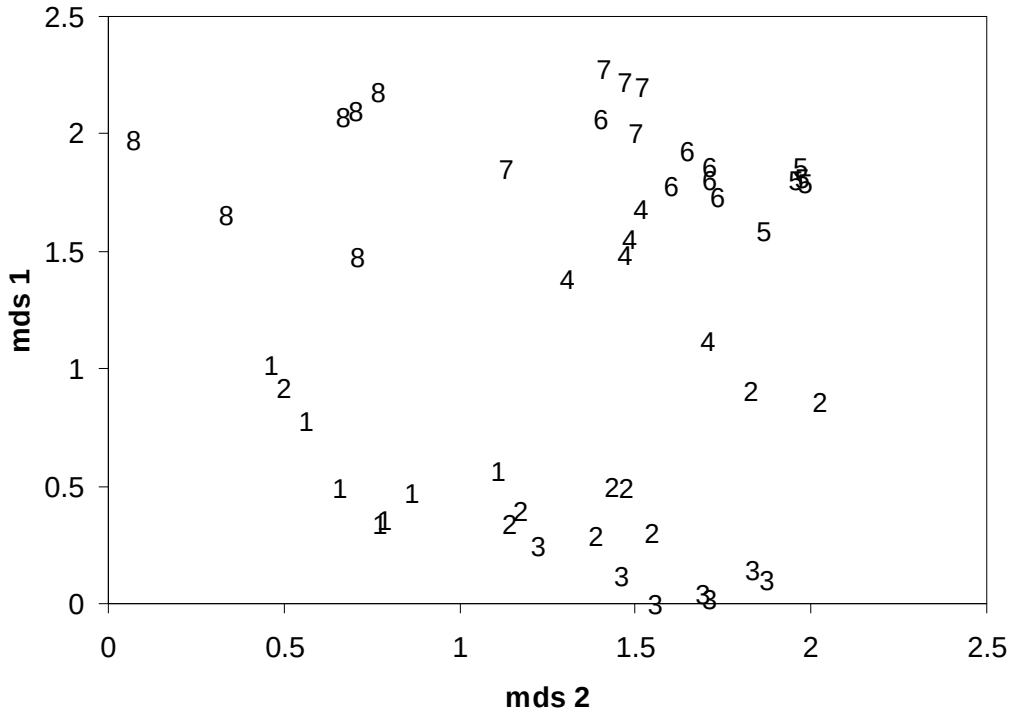

Figure 2.1: The distribution of communities in two dimensional ordination space. 1 = *Eucalyptus obliqua – Acacia verticillata – Olearia lirata – Blechnum nudum – Pteridium esculentum* open-forest. 2 = *Eucalyptus obliqua - Exocarpos cupressiformis - Olearia lirata - Pteridium esculentum* open-forest. 3 = *Eucalyptus obliqua - E. viminalis - Bedfordia salicina - Lomandra longifolia - Pteridium esculentum* open-forest. 4 = *Pteridium esculentum* fernland. 5 = *Senecio* spp. *- Holcus lanatus - Anthoxanthum odoratum – Acetosella vulgaris* grassland. 6 = *Holcus lanatus - Anthoxanthum odoratum - Bromus* spp. *grassland.* 7 = *Juncus pallidus - Holcus lanatus - Anthoxanthum odoratum - Acetosella vulgaris* rushland. 8 = *Carex appressa – Juncus* spp. sedgeland

In naming the communities the dominant species in the tallest stratum and the most common structural type (following Specht *et al*. 1974) were used. The community descriptions include a list of the plant taxa in more than 50% of the quadrats. These are grouped by life form. Note that the 'herb' life form includes all non-woody dicotyledonous vascular plants.

Floristic communities

1. *Eucalyptus obliqua – Acacia verticillata – Olearia lirata – Blechnum nudum – Pteridium esculentum* open-forest.

Trees: *Eucalyptus obliqua*
Shrubs: *Acacia verticillata, A. dealbata, Goodenia ovata, Olearia lirata, Melaleuca squarrosa*
Sedges: *Carex appressa*
Ferns: *Blechnum nudum, Pteridium esculentum*
Grasses: *Ehrharta stipoides*
Herbs: *Senecio* spp.*, Hydrocotyle hirta*
Graminoid: *Chiloglottis* spp.,

This community was defined by the presence of species in groups A, C, E and K, and by the absence of species in groups B and D. The quadrats in this community occurred on level ground in a gully between sandstone ridges. The canopy was dominated by *Eucalyptus obliqua* (stringybark) with *Acacia verticillata* (prickly mimosa) present in all quadrats and dominant in the understorey. It was distinguished from the other eucalypt communities (2 and 3) by the presence of species in groups K and A, which consist of *Carex appressa, Blechnum nudum* and other wet sclerophyll species. With the presence of wet sclerophyll species community 1 was subsequently less diverse in dry sclerophyll understorey species than communities 2 and 3, as indicated by the absence of species in groups B and C.

2. *Eucalyptus obliqua - Exocarpos cupressiformis - Olearia lirata - Pteridium esculentum* open-forest.
Trees: *Eucalyptus obliqua, Exocarpos cupressiformis*
Shrubs: *Olearia lirata*
Ferns: *Pteridium esculentum, Microsorum diversifolium*
Herbs: *Lagenifera stipitata, Stylidium graminifolium, Viola hederacea, Oxalis perennans, Senecio* spp.*, Geranium* spp., *Cerastium* spp.
Graminoids: *Lomandra longifolia, Chiloglottis* spp*. Luzula* spp.
Grasses: *Poa labillardierei, Ehrharta stipoides, Poa* spp.*, Aira* spp.*, Holcus lanatus*

Community 2 was defined by the presence of species from all groups with the exception of group G. It was dominated by *Eucalyptus obliqua* (stringybark) and had an understorey dominated by *Pteridium esculentum* (bracken). It also had a high diversity of sclerophyll shrubs such as *Oxylobium arborescens*, *Banksia marginata*, *Cassinia aculeata* and *Richea procera* (as indicated by group B in Table 2.1) that were not present in the other *Eucalyptus* communities. Quadrats from this community were located on moderate to steep slopes, predominantly on the slopes approaching the wetland, which explains the presence of *Melaleuca squarrosa* (scented paper-bark) and *Richea procera* in this community.

3. *Eucalyptus obliqua - E. viminalis - Bedfordia salicina - Lomandra longifolia - Pteridium esculentum* open-forest.
Trees: *Eucalyptus obliqua, E. viminalis, Exocarpos cupressiformis*
Shrubs: *Olearia lirata, Bedfordia salicina, Coprosma quadrifida.*
Ferns: *Pteridium esculentum*
Herbs: *Lagenifera stipitata, Geranium* spp., *Drosera peltata*
Graminoids: *Luzula* spp., *Lomandra longifolia, Chiloglottis gunnii, Chiloglottis* spp., *Corybas* spp.
Grasses: *Poa* spp., *Agrostis* spp.

Community 3, another *Eucalyptus* forest dominated by *E. obliqua* (stringybark), was distinguished by the absence of species from groups A and B. It was dominated by *Pteridium esculentum* (bracken) in the understorey along with a high diversity of sclerophyll shrubs and herbs. This community also had less exotic species than community 2, as shown by the lack of species from group H that occurred in community 2. Quadrats from community 3 had a wide distribution as they occurred on the moderate to gentle slopes that predominated on the western side of the property. The absence of species from group I indicates the lower number of exotic grasses and herbs found in this community compared to the other two forest communities.

4. *Pteridium esculentum* fernland.

Ferns: *Pteridium esculentum*

Herbs: *Hypochoeris radicata, Leontodon taraxacoides, Acetosella vulgaris*

Grasses: *Ehrharta stipoides, Aira* spp, *Holcus lanatus, Anthoxanthum odoratum*

Community 4 occurred on the plateau behind the western ridge and in the higher areas of the undulating ground of the northern pasture. As indicated by the name, *Pteridium esculentum* (bracken) was the dominant species with exotic grasses and herbs making up the ground cover. The introduced grasses *Holcus lanatus* (Yorkshire fog-grass) and *Anthoxanthum odoratum* (sweet vernal grass) were co-dominants in some of the quadrats sampled. This community was distinguished from the other pasture communities as it only included species in groups E and I due to the dominance of bracken. There was a very rare occurrence of two native species, *Gonocarpus teucrioides* and *Chiloglottis* spp., and when present, the native grass *Ehrharta stipoides* (weeping grass) was only represented by a very low (< 5%) cover.

5. *Senecio* spp. - *Holcus lanatus* - *Anthoxanthum odoratum – Acetosella vulgaris* grassland.

Herbs: *Senecio* spp., *Hydrocotyle hirta, Anagallis arvensis, Cirsium vulgare, Cerastium* spp., *Acetosella vulgaris, Trifolium repens, Trifolium* spp.

Grasses: *Poa* spp. *Danthonia* spp., *Holcus lanatus, Anthoxanthum odoratum*

Community 5 was dominated by the two exotic grasses, *Holcus lanatus* (Yorkshire fog-grass) and *Anthoxanthum odoratum* (sweet vernal grass), but was distinguished from the other grassland communities by the presence of *Senecio* spp. as a co-dominant. This community was characterised by species in groups E, F, I and J, but predominantly by groups F and I. Along with the presence of *Senecio* spp., the presence of the native herbs and grasses from group F distinguished this community from communities 4 and 7. This community occurred on the southern side of the pasture where the ground was higher than other pasture areas and had *Eucalyptus* open-forest (community 3) bordering its western side.

6. *Holcus lanatus - Anthoxanthum odoratum - Bromus* spp. grassland.
Graminoid: *Juncus* spp.
Herbs: *Acaena novae-zelandiae, Geranium* spp., *Senecio* spp., *Hydrocotyle hirta, Gnaphalium* spp., *Cirsium vulgare, Cerastium* spp., *Acetosella vulgaris, Trifolium repens, Trifolium* spp., *Cotula reptans*
Grasses: *Bromus* spp., *Holcus lanatus, Anthoxanthum odoratum*

This community was dominated by mixed stands of exotic grasses, with the addition of *Bromus* spp. to the common *Holcus lanatus* (Yorkshire fog-grass) and *Anthoxanthum odoratum* (sweet vernal grass) that predominated in the other grassland communities. It was characterised by species in groups F, H, I, J and K. Its major distinguishing feature was the presence of *Bromus* spp. as a co-dominant, as this taxon was rare or not present in the other grassland communities. The high diversity of group H species was also unique to this community. These species were native herbs and graminoids not detected in the other grassland communities. The location of this community was on the gentle slopes that occurred on the western side of the wetland where the bordering vegetation was *Eucalyptus* open forest and *Carex appressa* wetland.

7. *Juncus pallidus - Holcus lanatus - Anthoxanthum odoratum - Acetosella vulgaris* rushland.
Sedge: *Juncus* spp., *Juncus pallidus*
Herbs: *Hypochoeris radicata, Acaena novae-zelandiae, Geranium* spp., *Senecio* spp., *Hydrocotyle hirta, Gnaphalium* spp., *Acetosella vulgaris, Trifolium repens, Trifolium* spp.
Grasses: *Holcus lanatus, Anthoxanthum odoratum*

The ground cover of this community was also dominated by the exotic grasses *Holcus lanatus* (Yorkshire fog-grass) and *Anthoxanthum odoratum* (sweet vernal grass) along with *Trifolium repens* (white clover). However it was distinguished from the other grassland communities by the dominance of *Juncus* spp.. Community 7 was characterised by species in groups H, I and J. It was also distinguished from

communities 5 and 6 by the absence of group F species. As with the other grassland communities, this community occurred where the land was cleared and used as pasture but it was predominantly found on lower ground, in contrast to community 4. Community 7 occurred in patches on the north side of the pasture, which may have been influenced by the undulating ground on this side.

8. *Carex appressa – Juncus* spp. sedgeland
Sedge: *Carex appressa, Juncus* spp.,
Herbs: *Hydrocotyle pterocarpa, Acaena novae-zelandiae, Geranium* spp., *Senecio* spp., *Hydrocotyle hirta, Gnaphalium* spp.,
Grasses: *Holcus lanatus, Anthoxanthum odoratum*
Fern: *Azolla filiculoides*

Community 8 was characterised by species in groups D, E, F, G, I, J and K, but predominantly by species in groups I, J and K. The distinctive feature of this community was that it was dominated by *Carex appressa* and it bordered the wetland. Codominants included *Juncus* spp., along with the exotic grasses *Holcus lanatus* (Yorkshire fog-grass) and *Anthoxanthum odoratum* (sweet vernal grass). However the presence of the wetland species such as *Azolla filiculoides, Triglochin procera* and *Hydrocotyle pterocarpa* in mixed stands with *Carex appressa* made this community distinct. These species occurred on waterlogged soil (there was 1 cm to 2 cm standing water in some quadrats) that was present around the wetland. This community shared wetland species (J and K) with community 1 but was distinct in its lack of a tree stratum. *Melaleuca squarrosa* (scented paper-bark) did occur in this community but was sparse.

Vegetation types

Descriptions of the mapped vegetation communities (Figure 2.2) of the property are given below. The mapping units correspond closely to the described floristic communities with the addition of the wetland unit that was not sampled in the survey.

1. Wetland

1a. Surface - *Azolla filiculoides* and *Triglochin procera* aquatic herbfield.

The surface of the wetland was dominated by *Azolla filiculoides* and *Triglochin procera*, which was scattered throughout the wetland, predominantly in the shallower waters. A stand of *Eleocharis sphacelata* aquatic sedgeland occurred in the northeast of the wetland. *Myriophyllum* spp. were also common throughout. The water level of the wetland had been raised by the construction of a levy at the northwest corner. This northeast corner had water lilies covering the majority of the surface. This community resembles the statewide described *Triglochin procera* aquatic herbfield wetland community (Kirkpatrick and Harwood 1983).

1b. Margin – Sedgey *E. ovata* woodland/*Melaleuca squarrosa* closed-scrub

Community 8 dominated the margin of the wetland. However on the northern side there was a *Melaleuca squarrosa* dominated closed-scrub where *Acacia verticillata, Oxylobium arborescens* and *Richea procera* were also represented. Bracken and exotic grasses dominated the groundcover and a taller open forest stratum of scattered *Eucalyptus ovata* was present. This mapping unit also occurred in a small pocket on the southeast corner of the wetland. The quadrats that were sampled in this area belong to communities 1 and 2 because of the *Eucalyptus* tree stratum, though on the northeast side the vegetation was predominantly closed-scrub. This vegetation type is a mix between two statewide wetland communities, sedgey *E. ovata* woodland (Duncan and Brown 1985) and *Melaleuca squarrosa* scrub (Kirkpatrick and Harwood 1983).

2. *Eucalyptus* open-forest communities

2a. *Eucalyptus obliqua* open-forest

This mapping unit occurred in the damp area along the gully between the sandstone ridges. It corresponded with floristic group 1. *E. obliqua* dominated the tree stratum, which was of an open-forest density. In general it fits into a transition zone between shrubby *E. obliqua* forest and wet sclerophyll forest because of the co-occurrence of dry sclerophyll shrubs such as *Acacia verticillata* with wet sclerophyll species such as *Olearia lirata* and *Blechnum nudum* (Duncan and Brown 1985).

2b. Heathy *Eucalyptus obliqua/E. amygdalina* open-forest

This vegetation type corresponded with floristic community 2. The vegetation type was found on sandstone slopes. *E. amygdalina* was present as a subdominant. It had a multi-layered understorey dominated by *Exocarpos cupressiformis* (native cherry), *Olearia lirata* and a dense bracken ground layer. As its understorey contained wet sclerophyll elements along with species typical of heathy understories it corresponded closely to the statewide community of heathy *E. obliqua* forest (Duncan and Brown 1985). It is closely related to 2c but occurred on exposed rock outcrops and steeper slopes.

2c. Shrubby *E. obliqua* open-forest

This was the most widespread community at the Saltwater River site. The mapping unit corresponded largely with floristic group 3. *E. obliqua* dominated the tree stratum, with *E. amygdalina* (black peppermint) and *E. viminalis* (white-gum) occurring as sub-dominants. The shrub stratum was dominated by *Exocarpos cupressiformis* (native cherry), *Bedfordia salicina* and bracken. It also had a diverse ground cover that consisted of native herbs and graminoids. This vegetation type also had the least amount of exotics present. The statewide community classification of this mapping unit is shrubby siliceous *E. obliqua* forest (Duncan and Brown 1985).

3. Pasture zones

The pasture zones occur on the eastern side of the property and were classified into four mapping units. These mapping units correspond closely to the four grassland communities described from the sorted table.

3a. Bracken fernland

Mapping unit 3a, which corresponded with floristic community 4, was the large stand of *Pteridium esculentum* (bracken) fernland that occurred on the cleared plateau behind the western ridge. In this part of the pasture bracken dominated the tallest stratum with little or no occurrence of ground cover species.

3b. Grassland

This mixed grassland unit was found on the gentle slopes on the north-western side of the wetland. The most typical dominants were *Holcus lanatus* (Yorkshire fog-grass), *Anthoxanthum odoratum* (sweet vernal grass) and *Bromus* spp. *Juncus* spp. was also present throughout the majority of the mapping unit. The vegetation in this unit corresponded with floristic community 6. All of the quadrats sampled in this floristic group fell into this mapping unit.

3c. *Juncus* spp. rushland

This mapping unit was dominated by *Juncus* spp. in the upper stratum while exotic grasses and herbs dominated the ground layer. It was located on the low areas of the pasture where drainage was poor and corresponded with floristic community 7.

3d. *Senecio* spp. open-herbfield

This mapping unit corresponded with community 5 and occurred on the higher slopes of the southern side of the pasture. Introduced grasses and herbs dominated the ground cover while *Senecio* spp. dominated the upper stratum.

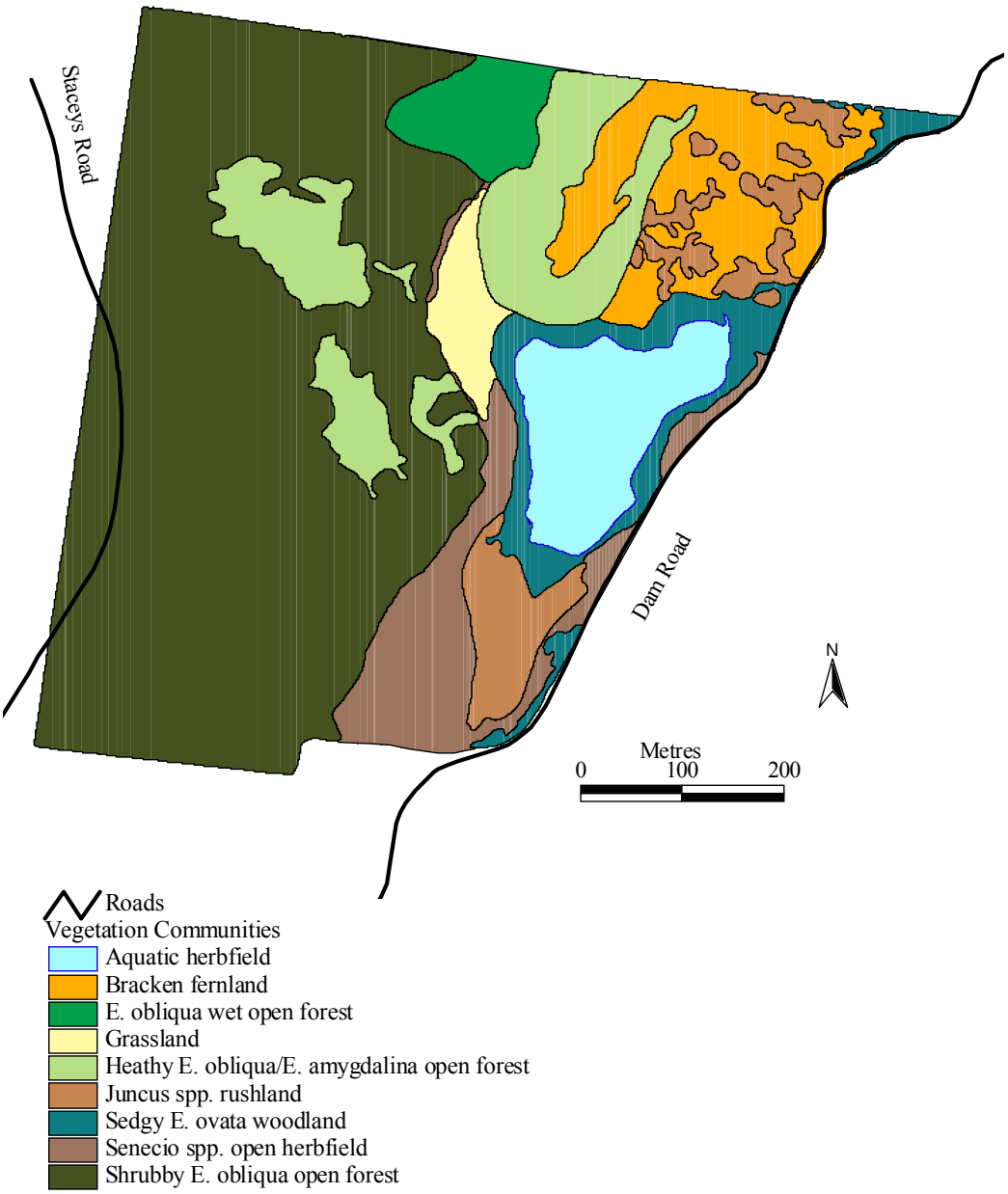

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Figure 2.2: The distribution of the present vegetation communities at the Saltwater River site.

Conservation status and species list of communities

Three communities that are mappable on a statewide basis were located on the property (Table 2.2). They were sedgey *E. ovata* woodland, shrubby *E. obliqua* forest and heathy *E. obliqua/E. amygdalina* forest. The conservation status of sedgey *E. ovata* woodland is extremely poor. Small stands are located in state reserves on the Tasman Peninsula including the Lime Bay Nature Reserve (Duncan and Brown 1985). In locations not subject to prolonged waterlogging *E. ovata* co-occurs with *E. amygdalina*. Shrub species density is high, except on sites which have been frequently fired and heavily grazed. At the study site *Melaleuca squarrosa* and *Acacia verticillata* dominate the medium shrub layer. When the shrub layer is more open, a comparatively tall, dense ground layer develops (Duncan and Brown 1985). Reconstruction of the sedgey *E. ovata* woodland community is recommended as it can be an important provider of shelter and breeding sites for certain bird and animal species (Harwood 1991).

The conservation value of shrubby *E. obliqua* forest is fair with unlogged stands of *E. obliqua* having a high conservation value. The eucalypts in this community are invariably mixed stands, containing two to several age classes. The proportion of peppermints increases as moisture availability and/or site fertility decreases. The shrubby *E. obliqua* community grades into wet sclerophyll as moisture availability increases, and the understorey contains both xerophytic and mesophytic (e.g. *Olearia, Bedfordia, Cassinia*) elements (Duncan and Brown 1985).

Heathy *E. amygdalina* forest on sandstone is found in south-eastern Tasmania where it is associated with Triassic sediments (mainly sandstone, where *E. obliqua* assumes dominance in gullies and shaded slopes). This community has a poor to moderate conservation status, as it occurs mainly on private land (Duncan and Brown 1985). The landform is undulating and altitudinal range is from sea level to 400 m. Soils are deep to moderately deep and drainage is free.

Table 2.2: Taxa of the three communities recommended for reconstruction.

Community	**Sedgey *E. ovata* woodland**	**Shrubby *E. obliqua* forest**	**Heathy *E. amygdalina* forest**
Trees	*Eucalyptus ovata*	*E. obliqua*	*Eucalyptus amygdalina*
	E. amygdalina	*E. amygdalina*	*E. obliqua*
	E. obliqua	*E. viminalis*	*E. viminalis*
	E. viminalis	*E. ovata*	*E. ovata*
			E. globulus
Shrub layer	*Melaleuca squarrosa*	*Acacia dealbata*	*Acacia dealbata*
	Oxylobium arborescens	*Bedfordia salicina*	*Acacia verticillata*
	Acacia verticillata	*Allocasuarina littoralis*	*Acacia melanoxylon*
	Acacia melanoxylon	*Exocarpos cupressiformis*	*Banksia marginata*
	Leptospermum lanigerum	*Monotoca glauca*	*Exocarpos cupressiformis*
	Banksia marginata	*Acacia melanoxylon*	*Leptospermum scoparium*
		Bursaria spinosa	*Melaleuca squarrosa*
		Olearia lirata	*Allocasuarina littoralis*
			Bursaria spinosa
			Olearia lirata
			Oxylobium arborescens

Low shrubs	*Aotus ericoides*	*Leptospermum scoparium*	*Epacris impressa*
	Astroloma humifusum	*Epacris impressa*	*Cassinia aculeata*
	Coprosoma quadrifida	*Aotus ericoides*	*Richea procera*
	Epacris impressa	*Pultenaea juniperina*	*Aotus ericoides*
	Gonocarpus teucrioides	*Pultenaea daphnoides*	*Bossiaea cinerea*
	Goodenia ovata	*Acacia ariceana*	*Pultenaea juniperina*
	Senecio quadridentatus	*Acacia dealbata*	*Astroloma humifusum*
		Amperea xiphoclada	*Leucopogon virgatus*
		Gonocarpus teucrioides	*Amperea xiphoclada*
		Gonocarpus tetragynus	*Lomatia tinctoria*
		Astroloma humifusum	*Monotoca glauca*
		Coprosma quadrifida	*Clematis aristata*
		Daviesia ulicifolia	*Daviesia ulicifolia*
		Senecio spp.	*Gonocarpus teucrioides*
		Goodenia ovata	*Pimelea drupacea*
		Goodenia lanata	*Persoonia juniperina*
		Pimelea drupacea	*Senecio* spp.

Ground layer	*Gahnia grandis*	*Pteridium esculentum*	*Pteridium esculentum*
	Lepidosperma elatius	*Hymenophyllum peltatum*	*Lomandra longifolia*
	Pteridium esculentum	*Blechnum nudum*	*Dianella tasmanica*
	Diplarrena moraea	*Polystichum proliferum*	*Diplarrena moraea*
	Lomandra longifolia	*Lomandra longifolia*	*Goodenia lanata*
	Poa labillardierei	*Diplarrena moraea*	*Stylidium graminifolium*
	Carex appressa	*Acaena novae-zelandiae*	*Viola hederacea*
	Juncus spp.	*Asplenium flabellifolium*	*Wahlenbergia* spp.
	Hypericum japonicum	*Billardiera longiflora*	*Acaena novae-zelandiae*
	Montia australasica	*Brachyscome spathulata*	*Billardiera longiflora*
	Schoenus spp.	*Cardamine species*	*Brachyscome spathulata*
	Centrolepis fascicularis	*Chiloglottis* spp.	*Chiloglottis* spp.
	Acaena novae-zelandiae	*Corybas* spp.	*Corybas* spp.
	Chiloglottis gunni	*Pterostylis* spp.	*Pterostylis* spp.
	Cotula reptans	*Viola hederacea*	*Stackhousia monogyna*
	Drosera peltata	*Stylidium graminifolium*	*Stipa* spp.
	Ehrharta stipoides	*Craspedia glauca*	*Drymophila cyanocarpa*
	Geranium spp.	*Danthonia* spp.	*Ehrharta stipoides*
	Drymophila cyanocarpa	*Drosera peltata*	*Euchiton* spp.
	Laginifera stipitata	*Euchiton* spp.	*Galium* spp.
	Hydrocotyle pterocarpa	*Galium* spp.	*Geranium* spp.
	Lobelia alata	*Geranium* spp.	*Helichrysum scorpioides*
	Picris spp.	*Lagenifera stipitata*	*Lagenifera stipitata*
		Luzula spp.	*Luzula* spp.
		Oxalis perennans	*Oxalis perennans*
		Pelagonium australe	*Poa labillardierei*
		Poa spp.	*Poa* spp.
			Picris spp.
			Poranthera microphylla

Vegetation reconstruction

The *Juncus* spp. mapping unit occurred on identical slopes to those within the *E. ovata* woodland mapping unit and the *E. obliqua* wet forest unit (Table 2.3). However the slopes were significantly different from those of the dry forest mapping units. Both the *Juncus* spp. and *E. ovata* mapping units were located on the poorly drained areas on the eastern side of the property (Figure 2.2). This corresponds with the information from Duncan and Brown (1985) who describe sedgey *E. ovata* woodland as distributed along drainage basins, valley flats, and in locations subject to seasonal waterlogging. Thus, in reconstructing the native vegetation (Figure 2.3) it can be construed that sedgey *E. ovata* woodland occurred in the areas where *Juncus* spp. now dominants.

Table 2.3: Results of the one-way ANOVA (Tukey's pairwise comparison, family error rate = 0.05) between floristic communities and slope. The * marks the floristic communities that occur on significantly similar slopes.

Floristic community	1	2	3	4	5	6	7
2	*						
3	*						
4							
5							
6							
7		*	*				
8		*	*				

The present distribution of the shrubby *E. obliqua* mapping unit is on the gentle slopes that predominate the property (Figure 2.2), which corresponds with its typical distribution in the foothills and uplands of ranges (Duncan and Brown 1985). The present distribution of the *Senecio* spp. mapping unit was on the south side and was bordered by shrubby *E. obliqua* open forest (Figure 2.2). Thus in reconstructing the native vegetation, shrubby *E. obliqua* open forest would be distributed in areas that are presently covered by the *Senecio* spp. mapping unit.

The heathy *E. obliqua/E. amygdalina* open-forest (Figure 2.2) was distributed on the steeper northern slopes and on the exposed rocky outcrops in the middle of the property. The cleared areas on the northern side were dominated by bracken. Grazing and frequent fires in heathy *Eucalyptus* forest (as had occurred on this property) increases the abundance of bracken in the low shrub layer (Duncan and Brown 1985). The distribution of the Bracken fernland can thus be used as an indication of the range of the heathy *Eucalyptus* forest before agricultural disturbance (Figure 2.3).

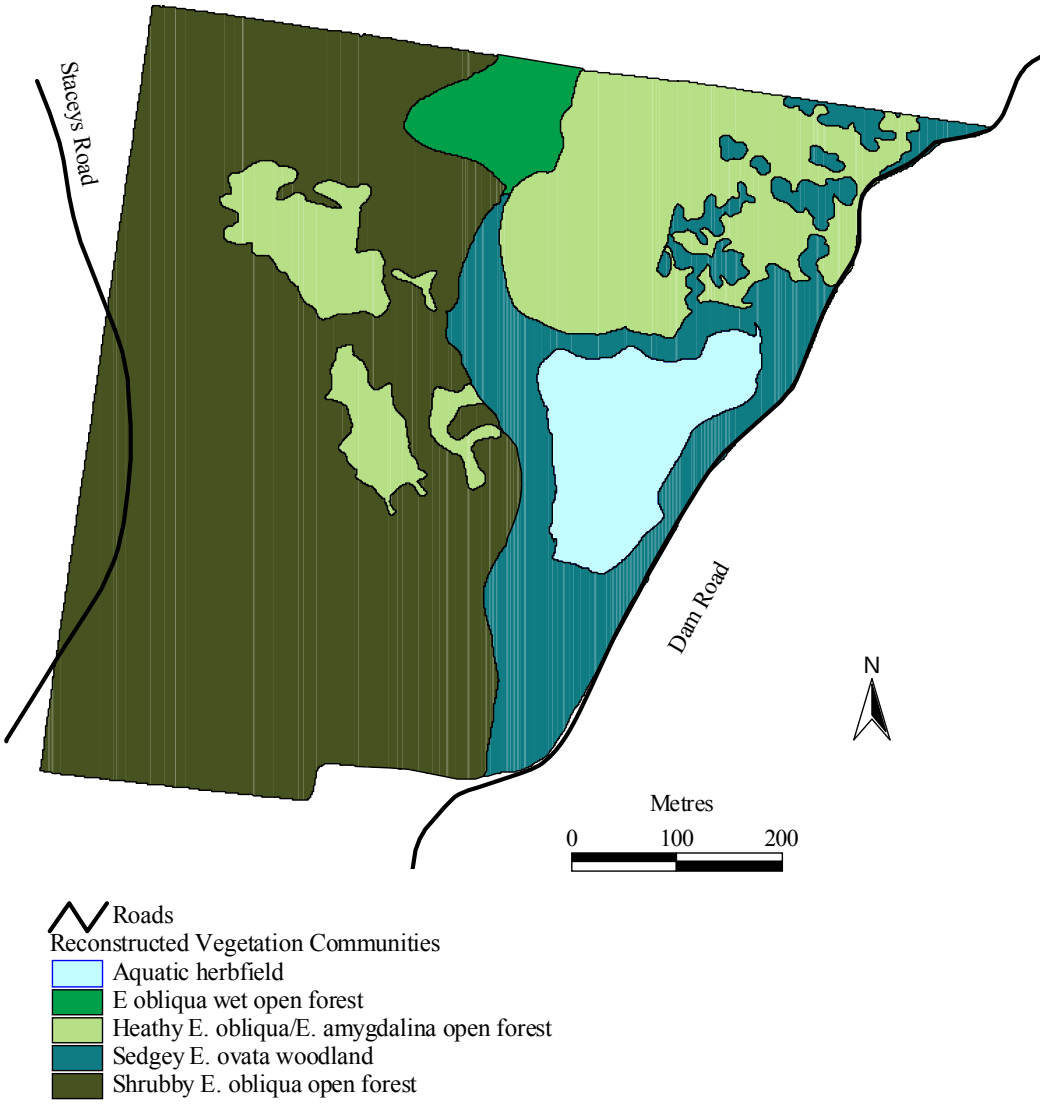


Figure 2.3: The distribution of the native vegetation communities at the Saltwater River site before agricultural disturbance.

Appendix 2.1: A list of all flora taxa that was identified during the 1999 survey of the study site. (i) = Introduced species, (e) = Tasmanian endemic, (o) = observed only, was not recorded in the quadrat sampling.

Family	Species name		Vegetation type
Apiaceae	*Hydrocotyle callicarpa*		Herb
	Hydrocotyle hirta		Herb
	Hydrocotyle muscosa		Herb
	Hydrocotyle pterocarpa		Herb
Aspleniaceae	*Asplenium flabellifolium*		Fern
	Asplenium obtusatum	o	Fern
Asteraceae	*Bedfordia salicina*		Shrub
	Brachyscome spathulata		Herb
	Cassinia aculeata		Shrub
	Cirsium vulgare	i	Herb
	Cotula coronopifolia	i/o	Herb
	Cotula reptans		Herb
	Craspedia glauca		Herb
	Euchiton spp.		Herb
	Helichrysum scorpioides		Herb
	Hypochoeris radicata	i	Herb
	Lagenifera stipitata		Herb
	Leontodon taraxacoides	i	Herb
	Olearia lirata		Shrub
	Picris spp.		Herb
	Senecio linearifolius		Herb
	Senecio quadridentatus		Herb
	Senecio spp.		Herb
	Sonchus asper	i/o	Herb
	Sonchus oleraceus	i/o	Herb
	Sonchus spp.	i	Herb
Azollaceae	*Azolla filiculoides*		Fern
Blechnaceae	*Blechnum nudum*		Fern
Brassicaceae	*Cardamine* spp.		Herb
Callitrichaceae	*Callitriche* spp.	o	Herb
Campanulaceae	*Lobelia alata*		Herb
	Lobelia spp.		Herb
Caryophyllaceae	*Cerastium glomeratum*	i	Herb
	Cerastium spp.	i	Herb
	Sagina spp.		Herb
	Spergularia spp.	i/o	Herb

	Stellaria media	i	Herb
Casuarinaceae	*Allocasuarina verticillata*	o	Shrub
Campanulaceae	*Wahlenbergia* spp.		Herb
Centrolepidaceae	*Centrolepis fascicularis*	o	Graminoid
Chenopodiaceae	*Einadia nutans*	o	Herb
Clusiaceae	*Hypericum japonicum*		Herb
Convolvulaceae	*Calystegia silvatica*	i	Herb
	Dichondra repens	o	Herb
Crassulaceae	*Crassula sieberana*		Herb
Cyperaceae	*Carex appressa*		Graminoid
	Eleocharis sphacelata	o	Graminoid
	Gahnia grandis		Graminoid
	Isolepis spp.		Graminoid
	Lepidosperma elatius		Graminoid
	Schoenus spp.		Graminoid
Dennstaedtiaceae	*Hypolepis rugosula*		Fern
	Pteridium esculentum		Fern
Dicksoniaceae	*Dicksonia antarctica*	o	Fern
Droseraceae	*Drosera peltata*		Herb
Dryopteridaceae	*Polystichum proliferum*		Fern
Ericaceae	*Astroloma humifusum*		Shrub
	Cyathodes glauca	o	Shrub
	Epacris impressa		Shrub
	Leucopogon virgatus	o	Shrub
	Monotoca glauca		Shrub
	Richea procera	e	Shrub
Euphorbiaceae	*Poranthera microphylla*		Herb
Fabaceae	*Aotus ericoides*		Shrub
	Daviesia ulicifolia		Shrub
	Lotus corniculatus	i/o	Herb
	Oxylobium arborescens		Shrub
	Pultenaea daphnoides		Shrub
	Pultenaea juniperina		Shrub
	Trifolium repens	i	Herb
	Trifolium spp.	i	Herb
	Vicia sativa	i	Herb
Gentianaceae	*Centaurium erythraea*	i	Herb
	Centaurium spp.	i	Herb
	Sebaea ovata		Herb
Geraniaceae	*Geranium* spp.		Herb
	Pelargonium australe		Herb

Goodeniaceae	*Goodenia lanata*		Herb
	Goodenia ovata		Shrub
Haloragaceae	*Gonocarpus micranthus*		Herb
	Gonocarpus tetragynus		Herb
	Gonocarpus teucrioides		Herb
	Myriophyllum spp.	o	Herb
Hymenophyllaceae	*Hymenophyllum peltatum*		Fern
Iridaceae	*Diplarrena moraea*		Graminoid
Juncaceae	*Juncus bufonius*		Graminoid
	Juncus pallidus		Graminoid
	Juncus pauciflorus		Graminoid
	Juncus spp.		Graminoid
	Luzula flaccida	o	Graminoid
	Luzula spp.		Graminoid
Juncaginaceae	*Triglochin procera*		Graminoid
Liliaceae	*Dianella tasmanica*		Graminoid
	Drymophila cyanocarpa		Graminoid
Mimosaceae	*Acacia dealbata*		Shrub
	Acacia melanoxylon		Tree
	Acacia verticillata		Shrub
Myrtaceae	*Eucalyptus amygdalina*		Tree
	Eucalyptus globulus	o	Tree
	Eucalyptus obliqua		Tree
	Eucalyptus ovata		Tree
	Eucalyptus viminalis		Tree
	Leptospermum lanigerum		Shrub
	Melaleuca squarrosa		Shrub
Onagraceae	*Epilobium* spp.		Herb
Orchidaceae	*Chiloglottis gunnii*	e	Graminoid
	Chiloglottis spp.		Graminoid
	Corybas spp.		Graminoid
	Pterostylis pedunculata		Graminoid
	Pterostylis spp.		Graminoid
Oxaliaceae	*Oxalis perennans*		Herb
	Oxalis pes-caprae	i/o	Herb
	Oxalis purpurea	i/o	Herb
Pittosporaceae	*Billardiera longiflora*		Climber
	Bursaria spinosa		Shrub
Plantaginaceae	*Plantago lanceolata*	i	Herb
Poaceae	*Agrostis capillaris*	i	Grass
	Agrostis spp.	i	Grass

	Agrostis stolonifera	i	Grass
	Aira spp.	i	Grass
	Anthoxanthum odoratum	i	Grass
	Bromus spp.	i	Grass
	Dactylis glomerata	i/o	Grass
	Danthonia spp.		Grass
	Ehrharta stipoides		Grass
	Holcus lanatus	i	Grass
	Poa annua	i/o	Grass
	Poa labillardierei		Grass
	Poa sieberiana	o	Grass
	Poa spp.		Grass
	Stipa spp.		Grass
	Vulpia spp.	i	Grass
Polygalaceae	*Comesperma volubile*	o	Shrub
Polygonaceae	*Acetosella vulgaris*	i	Herb
Polypodiaceae	*Microsorum diversifolium*		Fern
	Rumex brownii		Herb
Portulacaceae	*Montia australasica*		Herb
Primulaceae	*Anagallis arvensis*	i	Herb
Proteaceae	*Banksia marginata*		Shrub
	Lomatia tinctoria	e	Shrub
	Persoonia juniperina		Shrub
Pteridaceae	*Pteris tremula*	o	Fern
Ranunculaceae	*Clematis aristata*		Climber
	Ranunculus repens	i/o	Herb
	Ranunculus glabrifolius	o	Herb
	Ranunculus lappaceus	o	Herb
	Ranunculus spp.		Herb
Rosaceae	*Acaena novae-zelandiae*		Herb
	Acaena echinata	o	Herb
	Aphanes arvensis	i/o	Herb
	Rubus fruticosus	i	Climber
	Rubus spp.	i	Climber
Rubiaceae	*Coprosma quadrifida*		Shrub
	Galium spp.		Herb
Rutaceae	*Zieria arborescens*		Shrub
Santalaceae	*Exocarpos cupressiformis*		Shrub
	Leptomeria drupacea		Shrub
Scrophulariaceae	*Euphrasia collina*		Herb
	Euphrasia spp.		Herb

	Mazus pumilio		Herb
Solanaceae	*Solanum* spp.	o	Herb
Stackhousiaceae	*Stackhousia monogyna*		Herb
Stylidiaceae	*Stylidium graminifolium*		Herb
Thymelaeaceae	*Pimelea drupacea*		Shrub
	Pimelea humilis	o	Shrub
Violaceae	*Viola hederacea*		Herb
Xanthorrhoeaceae	*Lomandra longifolia*		Graminoid

Chapter Three

Examination of factors influencing native seedling emergence

Introduction

As the natural recovery of native vegetation on ex-arable sites can take several decades, it is important to understand the natural successional processes in abandoned pastures to help identify the factors most important in limiting recovery (Hutchings and Booth 1996; Holl *et al*. 2000; Buisson and Dutoit 2004). An understanding of the multiple factors that may retard recovery, such as lack of seed dispersal, nutrient limitation, water stress, and weed competition is required to design effective restoration strategies (Holl *et al.* 2000).

A vegetation survey of the study site (Chapter 2) showed that the modified pastoral zones were mostly improved pasture dominated by exotic grass species, particularly in the south-eastern part of the site, with little to no native recovery. Insufficient seed dispersal is a major limitation to natural regeneration at many neotropical sites (Zimmerman *et al*. 2000; Hooper *et al*. 2005) and many studies in Mediterranean-type ecosystems have reported that native seed dispersal into abandoned agricultural lands is limited due to the lack of available native propagules (Grubb and Hopkins 1986; Zeng and Whelan 1993; McDonald 1996). Native species of south-eastern Australia that have been found to establish in improved pastures have hard seeds that can persist in the soil (Onans and Parsons 1980) or adaptations that assist in dispersal over long distances. Appleby (1998) found that re-establishment of native vegetation on improved pastures in south-eastern Tasmania was generally restricted to areas adjacent to undisturbed native forest, with the recovery of the native community depending on migration of propagules.

The recolonisation of native temperate *Eucalyptus* woodland and forest species onto abandoned improved pastures has also been found to be often impeded by the competitive introduced grass sward (Maron and Connors 1996; Parrotta *et al*. 1997; Withers 1978; Yates and Hobbs 1997a; Lamb 1998). Fensham and Kirkpatrick (1992) found that the primary reason *Eucalyptus* seedlings failed to establish in dense grassy vegetation was due to competition for moisture and root space. However, eucalypts near the edge of a pasture have the potential to inhibit exotic grasses by depleting available nutrients in the soil, increasing soil acidity, or by the release of allelopathic compounds (Attiwill 1980; Kirkpatrick 1997). Ellis (1992) observed that, in dry forests of low elevations in Tasmania, and provided that browsing by native animals is at a low level, local eucalypts usually establish readily from seed in both native and introduced pasture.

Gaps in the grass sward that are created by disturbances such as animal diggings, grazing and fire may facilitate the re-establishment of remnant vegetation (McConnaughay and Bazzaz 1990; Reader and Buck 1991; Vieira *et al*. 1994; Belsky and Blumenthal 1997) and trial disturbances of soil seed banks can help to predict whether an assisted regeneration approach is needed on a restoration site (McDonald 1996). Cione *et al*. (2002) found that the application of grass-specific herbicide and hand cultivation resulted in successful native shrub establishment in exotic grassland communities of southern California, with continued efforts needed to minimise the reinvasion of exotic grasses, otherwise restoration efforts would be reversed.

Although gaps can create openings for native establishment, small gaps are often revegetated rapidly by weedy species (Hobbs and Huenneke 1992; Toth *et al*. 1993), whereas larger gaps created by severe disturbances such as fire or intense grazing may provide more suitable conditions for invasion (Christensen and Burrows 1986). The restoration of eucalypts and other woody species that recruit after natural disturbance events may require the application of techniques that mimic this disturbance, such as fire to stimulate the germination of fire-adapted species (Rokich *et al*. 2002). However, soil in large gaps free of litter can be subject to alternating cycles of wetting and drying which can be detrimental to young seedlings (Battaglia and Reid 1993).

Grazing may provide opportunities for some species but can prevent the establishment of others, with Fensham and Kirkpatrick (1992) finding that though heavy grazing can result in the loss of less tolerant species and increases in exotic species, protection from grazing on high nutrient sites can exclude native species. In their examination of the restoration of woody pastures on former agricultural land, Van Uytvanck *et al.* (2008) concluded that moderate grazing induces the development of diverse vegetation but having an initial time gap before grazing may enhance regeneration success of palatable tree seedlings. In temperate Australian environments Dorrough and Moxham (2005) observed that the highest probability of native woody plant regeneration on agricultural land was found on ungrazed sites. However, regeneration was also found to occur under intermittent grazing.

In improved pastures where the soil seed bank of native species may be depleted, re-establishment of native species may require direct seeding to initiate restoration (Venning 1988) and where natural regeneration is poor the sowing of woody plants should be used to increase plant density and diversity (Slocum *et al.* 2006). Yates and Hobbs (1997b) found that the restoration of *Eucalyptus salmonophloia* woodlands on agricultural land dominated by introduced annuals was only possible with a range of interventions, including supplementation of propagule availability by direct seeding or planting.

Direct broadcast seeding is currently the preferred revegetation practice to restore eucalypt forest after bauxite mining (McChesney *et al*. 1995). However, the germination requirements, such as the effects of dark/diurnal light, for many plant species in Australian temperate communities is poorly understood (Clarke *et al*. 2000). Some dominant sclerophyll tree and shrub species have been found to have reduced germination in light, with larger seeded species less sensitive to exposure to light than smaller seeded species (Bell *et al*. 1995; Facelli and Ladd 1996). However, laboratory trials do not necessary reflect the results obtained under field conditions, for example Clarke *et al*. (2000) and, Clarke and Davison (2001) found that under laboratory

conditions, more *Allocasuarina littoralis* seed germinated in the light whereas in the field greater germination occurred when seeds were buried.

The aim of this chapter is to examine the factors which potentially influence seedling recruitment of native forest vegetation in the improved pasture zones of the study site and how this information can be used to facilitate ecosystem recovery. A field experiment was conducted to examine seedling emergence of four tree and shrub species, *Eucalyptus obliqua, Allocasuarina littoralis, Acacia melanoxylon* and *A. verticillata*, typical of the remnant forest bordering the pasture. Emergence was examined in terms of:

i. distance from native forest
ii. grazing
iii. gaps created by soil disturbance to mimic animal diggings
iv. weed removal
v. burial of seeds

Materials and methods

Study area and ground preparation

An area on the south-eastern side of the pasture was chosen for this experiment as it had the least interference from forest boundaries on more than one side. The native forest that bordered the experimental area was shrubby *Eucalyptus obliqua* open forest and the pasture community that encompassed the experimental plots was the *Senecio* spp. open herbfield vegetation community (see Chapter 2). The experiment was split into four blocks, two fenced areas, constructed 1 year prior to the field trial, to exclude wallabies, rabbits and possums, and two unfenced areas as shown in Figure 3.1. Each block was 8 m wide and 45 m long, and included ten rows of plots, with the distances between rows increasing exponentially in distance from the forest edge (see Table 3.1).

Figure 3.1: Experimental plots that were contained in the four blocks (2 fenced and 2 unfenced) in the *Senecio* spp. open herbfield vegetation community that was bordered by shrubby *Eucalyptus obliqua* open forest on the western side. Photo was taken at the end of the field trial. Note that the area to the left of the south fence was fenced for a different experiment.

Table 3.1: The distances of experimental rows from the forest edge.

Row number	Distance of plot from forest edge (m)	Distance from previous row (m)
1	1.5-2.5	n.a.
2	3.1-4.1	0.6
3	5.3-6.3	1.2
4	8.1-9.1	1.8
5	11.5-12.5	2.4
6	15.5-16.5	3.0
7	20.1-21.1	3.6
8	25.3-26.3	4.2
9	31.1-32.1	4.8
10	37.5-38.5	5.4

Each row within each block was comprised of three 1 m^2 plots, with a minimum distance of 1 m between plots and fence edges. In each of the rows (of each block) there were 2 treated plots and one control plot. The treated plots were split into two halves perpendicular to the forest boundary. Each half was then randomly selected to have one of the two ground preparation treatments applied, which were soil disturbance and weed removal. Thus within one row of each block there were two plots with both ground preparation treatments applied to an rectangular area with the dimensions 1 m × 0.50 m. This method of application was chosen as the treated plots had either a soil disturbance or fire treatment applied to them 8 months previously, so to ensure this previous application of treatments did not affect the outcome of the current experiment both the previous fire and soil disturbance plots had the current ground preparation treatments applied to one half of each of the plots. The soil disturbance treatment comprised of turning over the turf with a hoe to mimic natural animal diggings while weed removal involved removing all of the turf along with root matter to a depth of 5 cm with a mattock.

Germination trials

Seed from four native plant species were used in the experiment. These were *Eucalyptus obliqua*, *Allocasuarina littoralis*, *Acacia melanoxylon* and *A. verticillata.* The *Eucalyptus* and *Allocasuarina* seeds were collected along Dam Rd, within 10 km of

the field site using secateurs and long handled pruners. The *E. obliqua* seeds were collected in February 2001 as *E. obliqua* seed needs to be collected during the summer (Ralph 1994). As *A. littoralis* holds seed all year round, its seeds were collected just prior to sowing in August 2001. The *Acacia* seed was purchased from 'Wildseed' who collected the seed from native forest on the Tasman Peninsula from November 2000 to March 2001. Prior to sowing, *Acacia* seeds were heat treated with boiling water for 5 minutes (Langkamp 1987; Ralph 1994; Bell 1999).

Laboratory trial

To test the viability of the seed, a laboratory trial was conducted using a growth cabinet programmed with 12 hours light a day and at a temperature of 25^0 Celsius. Twenty–five seeds of each species were placed in a petri dish containing moist litmus paper. Five petri dishes of each species were then placed in the diurnal environment while another five petri dishes were covered in foil to mimic light conditions when seeds are buried. The petri dishes were placed in the growth cabinet using a randomised block design with shelves of the growth cabinet treated as blocks. The petri dishes were checked daily to record any germination of seeds and the litmus paper was moistened if dry. The petri dishes of the dark treatment were only inspected under green light and rewrapped in foil following inspection.

Field trial

At the end of September 2001 seeds from the four species were sown at the study site. Each of the 1 × 0.50 m ground treated areas was split again into four smaller rectangular areas with the dimension of 25 cm × 50 cm. The following sowing treatments were then randomly applied to each ground treatment (i.e. each ground treatment had 4 subplots, thus treatment quadrats contained 8 subplots of treatments):

- surface sown seeds,
- buried seeds,
- no seeds (in two areas),

As control plots didn't have any ground preparation, only the surface sown seeds were applied to these plots. As with the treated plots, the sowing treatment in the control plots was applied to a 25 × 50 cm section that was randomly selected, and another randomly selected subplot of the same dimensions was used to record in situ native plant germination. Sowing treatments were comprised of 25 seeds of each species. *Acacia* and *Allocasuarina* seeds were counted whereas weight was used to estimate the number of *Eucalyptus* seeds due to their small size. Seedling emergence was recorded weekly for the first 8 weeks, then at weeks 10 and 20. To measure cumulative germination over the 10 weeks, individual seedlings were marked with pins so newly emerged seedlings could be counted in addition to any seedlings that perished since the last emergence count.

Environmental variables

The slope and aspect of the blocks were recorded. Soil moisture was measured in each row in week 8 and twice in week 10 (morning and evening) of the field trial using a Time Domain Reflectometry (TDR) soil moisture probe. The bare ground of each subplot was measured at weeks 4 and 10.

Soil samples were randomly collected from non-treatment areas at each row within each block in September 2001. The soil cores were 10 cm deep and 5 cm in diameter. The samples were air-dried and then coarsely sieved to remove roots, litter and stones. The following chemical analyses were completed:

- available phosphorus using Bray extractable method (Jackson 1958);
- total phosphorus using the Perchloric/Nitric/Sulphuric acid digestion method (Allen 1974);
- total nitrogen using the Kjeldahl method (Jackson 1958);
- total organic carbon using the Walkley and Black method (Rayment and Higginson 1992).

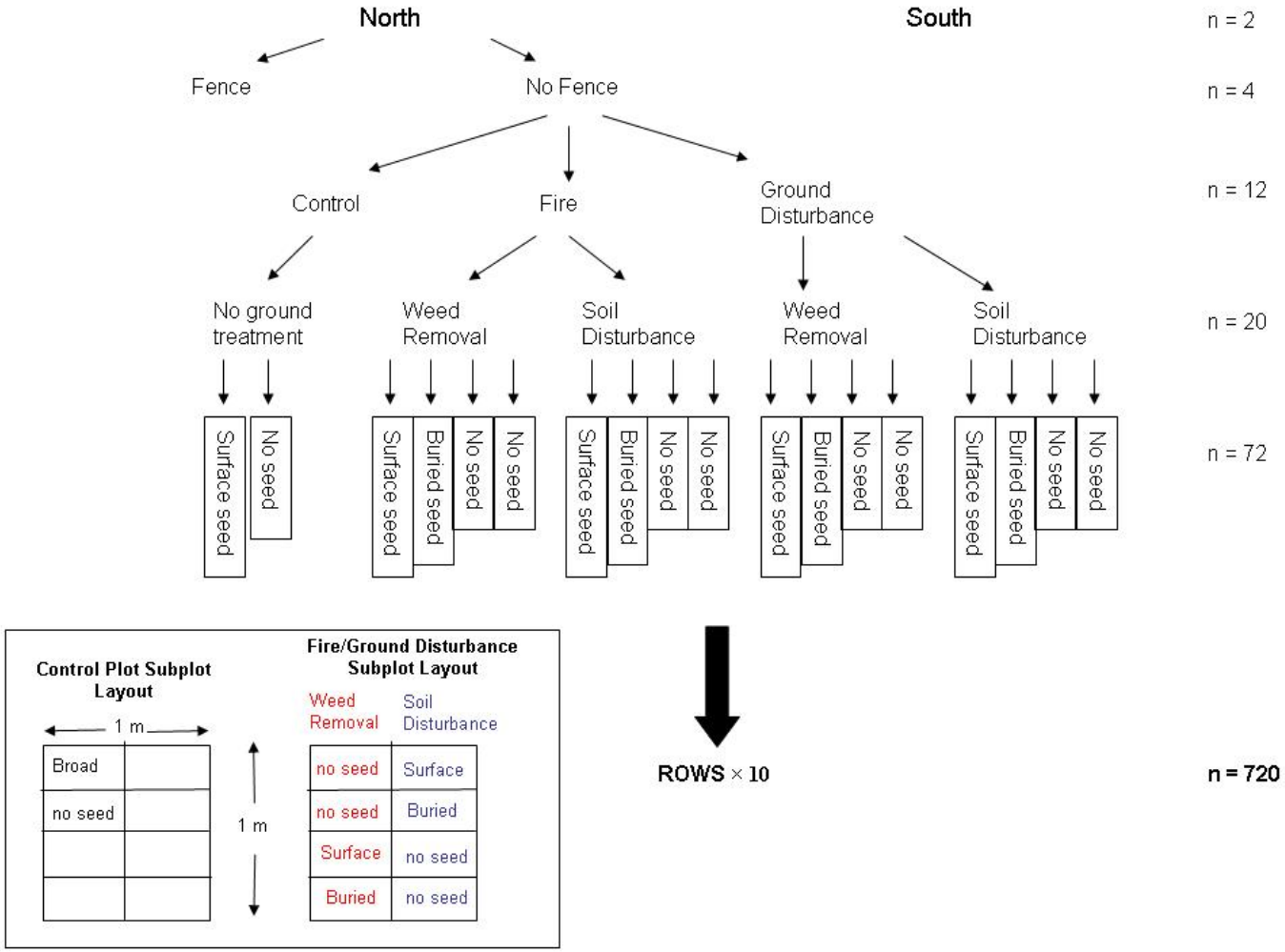


Figure 3.2: The factorial design of the field trial that included interactive effects of location, fencing, plot history, ground treatment, sowing treatment and distance from forest. The inset box shows the layout of sub-plots within the both the control and treated plots.

Analysis

Soil properties and bare ground cover

The combination of location and fencing on the variation in soil moisture and soil chemistry was analysed using the Kruskal-Wallis test. If no significant effect was detected across the two factors then data were pooled to analyse (using the Kruskal-Wallis test) the effect of distance from the forest edge on soil properties. Where a significant difference was detected by the Kruskal-Wallis test, the Mann-Whitney U test was used to test differences between two samples.

The effects of block, fencing, ground preparation and distance from forest on the percentage of bare ground covering subplots after 10 weeks was also analysed using the Kruskal-Wallis test. Post-hoc comparisons were undertaken using the Mann-Whitney U test (Sokal and Rohlf 1995; Dytham 1999).

Laboratory trial

Data from the viability tests were analysed for each of the four species using the chi-square test of association ($p < 0.05$) to determine if the proportion of seeds that germinated varied across the dark and diurnal conditions.

Field trial

Initially, percentage germination of seeds for each species under the different treatments was calculated, however, as the majority of plots yielded little to no seedling emergence, there was insufficient data to statistically compare the means (calculated from percentage germination) of the different treatments. Therefore for statistical analysis, seeding emergence observations were assigned to one of two categories, plots with germinants and plots without germinants. Furthermore, due to poor emergence, the experimental design became unbalanced which meant that the independent variables and their interactions could not be considered simultaneously in a multi-factorial analysis Thus the cumulative proportion of plots in which seedling emergence was recorded was analysed using chi-square tests for each level of experimental factors (in descending

order as outlined in Figure 3.2) for each of the sown species. Where plot effects were not evident for a particular factor ($p > 0.05$), data were pooled when analysing the effect of other treatment factors (i.e. if there was no significant effect of experimental blocks, data were pooled across the two blocks to test the effect of fencing on seed germination).

Results

Soil properties and bare ground cover

Soil moisture and chemistry

The experimental area had a slight downward slope of 1 degree from the south to the north side, whereas the slope from the forest edge to the east was a downward slope of 12 degrees. Soil moisture did not significantly vary across the four blocks ($p = 0.618$) but was found to significantly vary between rows, in the west to east direction ($p < 0.0005$). As distance increased from the forest edge so did soil moisture (Figure 3.3).

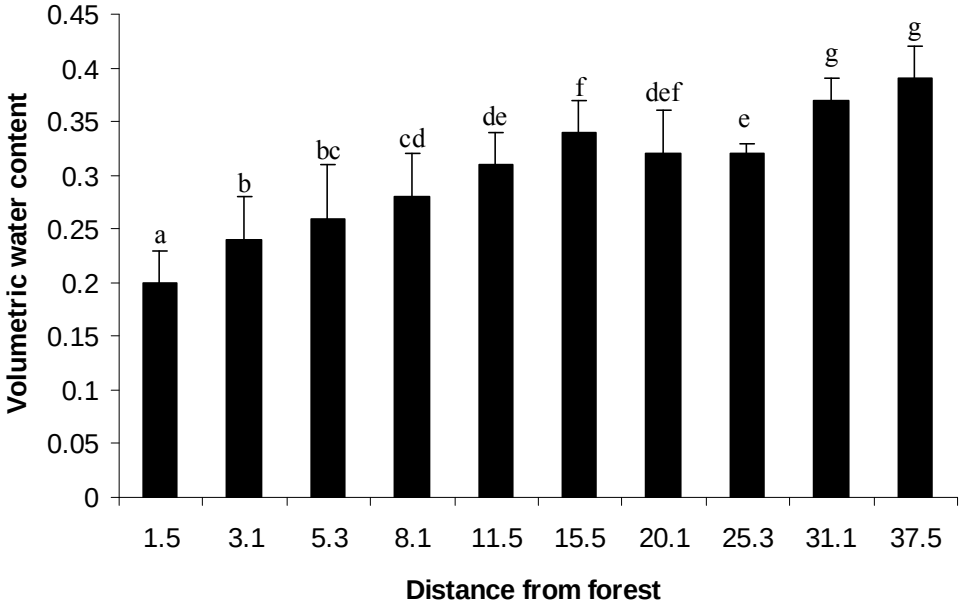


Figure 3.3: Mean soil moisture in relation to distance from the remnant forest (Error bars = standard error). Means were calculated from data measured on 3 occasions and pooled across four blocks (n = 12). Columns labelled with different letters were found to be significantly different at $p < 0.05$ (Mann-Whitney U test).

Percentage nitrogen, available phosphorus and the carbon/nitrogen ratio, did not vary across the two fenced and unfenced blocks or with distance from forest edge (Figures 3.4 and 3.5). However the percentage of carbon in the soil was found to vary across the four experimental blocks (Figure 3.6), with the south unfenced area recording a significantly higher percentage of carbon than the two fenced areas.

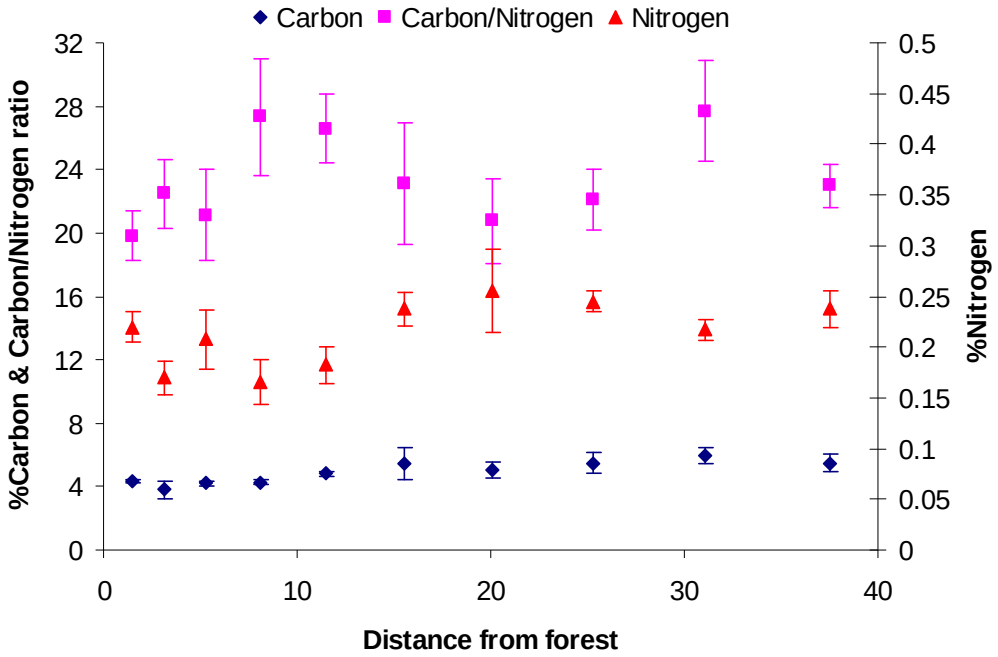


Figure 3.4: Mean soil nutrient levels in the soil in relation to distance from the remnant forest (Error bars = standard error). Note: two y-axes used. %Nitrogen data points are read using the right hand y-axis whereas the %Carbon and C/N data points are read using the left hand y-axis.

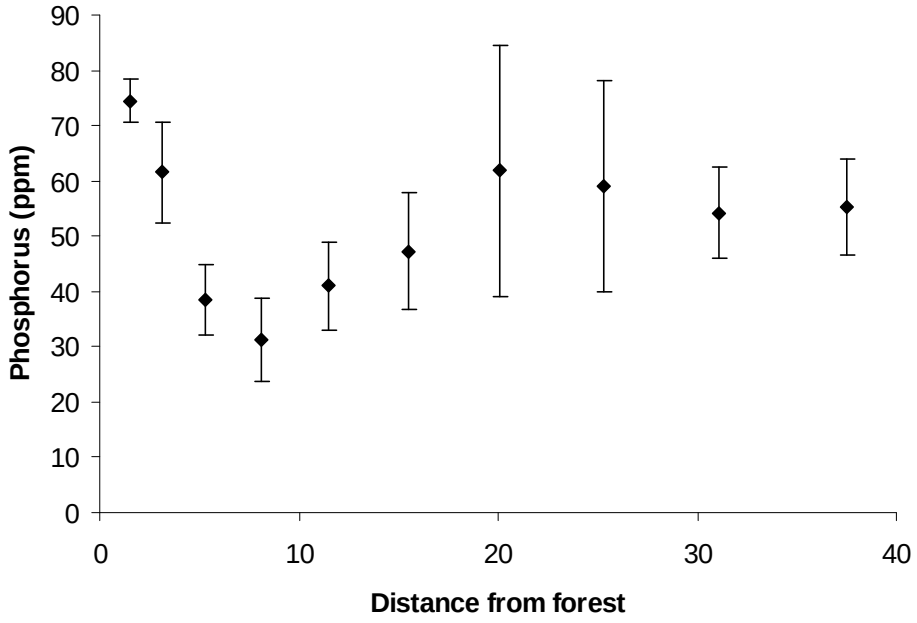


Figure 3.5: Mean available phosphorus levels in the soil in relation to distance from the remnant forest (Error bars = standard error).

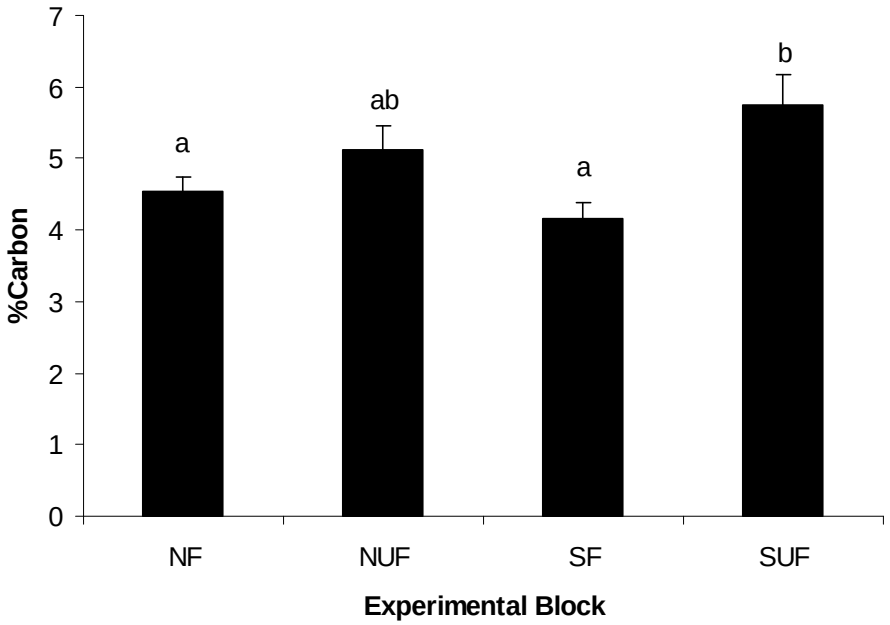


Figure 3.6: Mean percentage soil carbon across four experimental blocks (Error bars = standard error, n =10). Columns labelled with different letters were found to be significantly different at $p < 0.05$ (Mann-Whitney U test). N = North, S = South, F = Fenced, UF = Unfenced.

Bare ground cover

Bare ground cover was found to be significantly affected by the fencing treatment ($p = 0.0027$, Mann-Whitney U test) but not by the north or south position of the blocks. The mean amount of bare ground in fenced plots was 42 ± 1.4 % while 36 ± 1.4 % was the mean bare ground cover of unfenced plots. However, within both the fenced and unfenced areas, distance from forest edge did not have a significant effect on the amount of bare ground.

Control plots in both fenced and unfenced areas had negligible bare ground cover, which was significantly less than the bare ground cover of plots which had the ground treatment applications of either soil disturbance or weed removal. Soil disturbance resulted in significantly less bare ground cover 10 weeks after application than the treatment of weed removal in both fenced ($p < 0.0005$) and unfenced areas ($p < 0.0005$). There was no significant difference between the mean percentage bare ground of fenced

and unfenced soil disturbed plots, although fencing did result in significantly less bare ground covering plots which had the weed removal treatment (Figure 3.7).

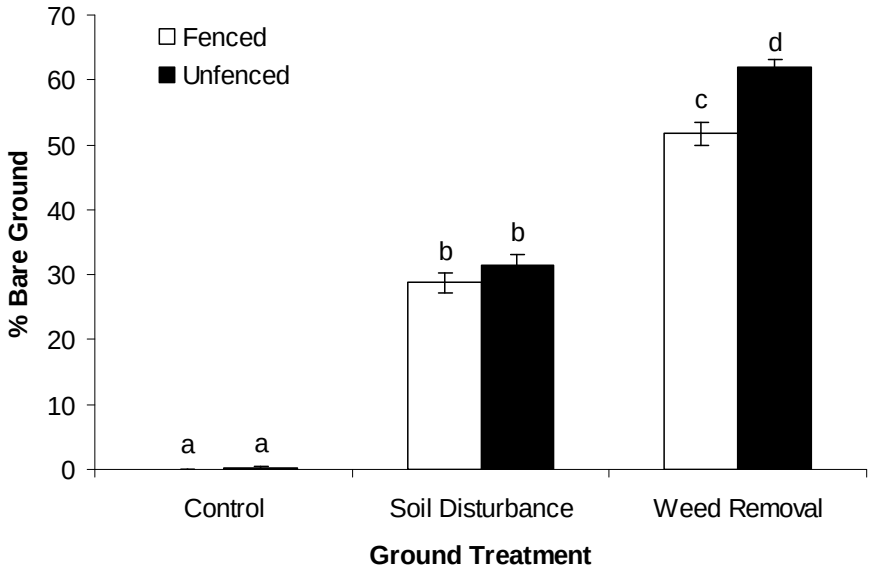


Figure 3.7: The mean percentage bare ground cover recorded in ground treated and non treated plots in both fenced and unfenced areas (Error bars = standard error). Columns labelled with different letters were found to be significantly different at $p < 0.05$ (Mann-Whitney U test).

Laboratory trial

Seed from both *Acacia* species showed high germinability in both the diurnal and dark treatments. *Acacia verticillata* had 100% germination in the diurnal conditions which was significantly greater than the percentage germination of 88 ± 8% in the dark conditions. *Acacia melanoxylon* had equally 91% germination in both conditions. *Allocasuarina littoralis* had similar germinability in both conditions with 33% in the diurnal conditions and 29% when seeds had no light source. *E. obliqua* seeds showed the poorest germinability with only 5.6 ± 3% germinating in the diurnal conditions which was significantly different to the percentage germination of 12.8 ± 2% that occurred in the dark conditions (Figure 3.8).

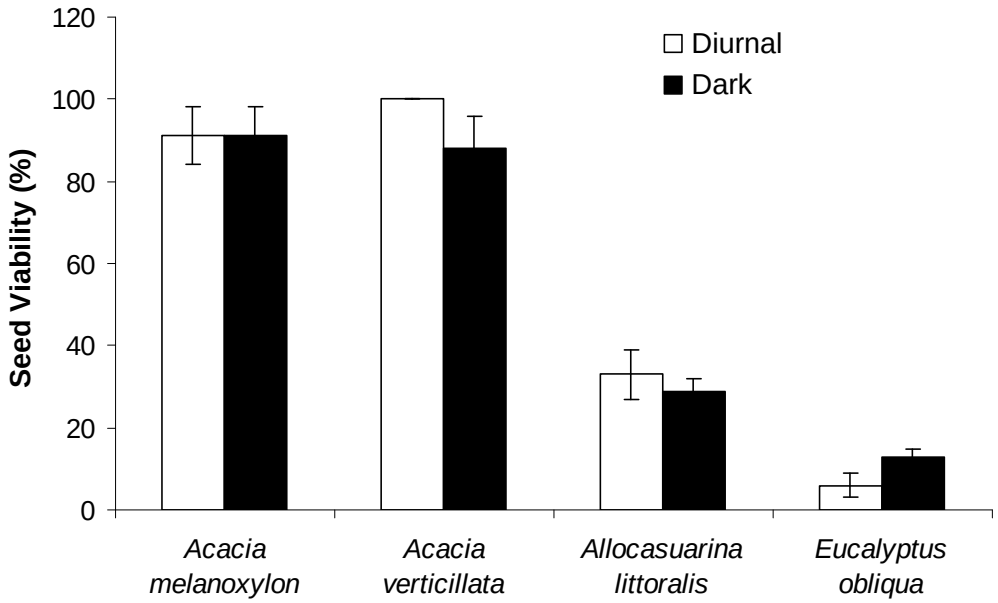


Figure 3.8: Seed germinability results for four native plant species. Data points indicate mean ± standard error, n = 5, n = 3 for *Acacia melanoxylon*.

Field trial

Seedlings started to emerge 4 weeks after sowing for *Allocasuarina littoralis* and *E. obliqua* and 5 weeks after sowing for both *Acacia* species (Figure 3.9). *A. melanoxylon* had the greatest number of seeds which germinated in the field trial, with 585 (6.5 % of sown seed) seeds germinating in 136 (38 %) of the 360 sown subplots. Five hundred and eleven *A. verticillata* seeds germinated (5.6 % of sown seeds) in 115 (32 %) sown subplots. A cumulative total of 80 (0.9 %) *Allocasuarina* seeds germinated in 58 (16 %) sown subplots while only 51 (0.6 %) *E. obliqua* seeds germinated in 38 (10.6 %) subplots. There was no emergence of any of the four species in subplots (total = 360) with unsown surfaces. For each of the fours species cumulative germination essentially peaked at 10 weeks, with only 2 *A. melanoxylon* seedlings and 4 *A. verticillata* seedlings emerging at 20 weeks.

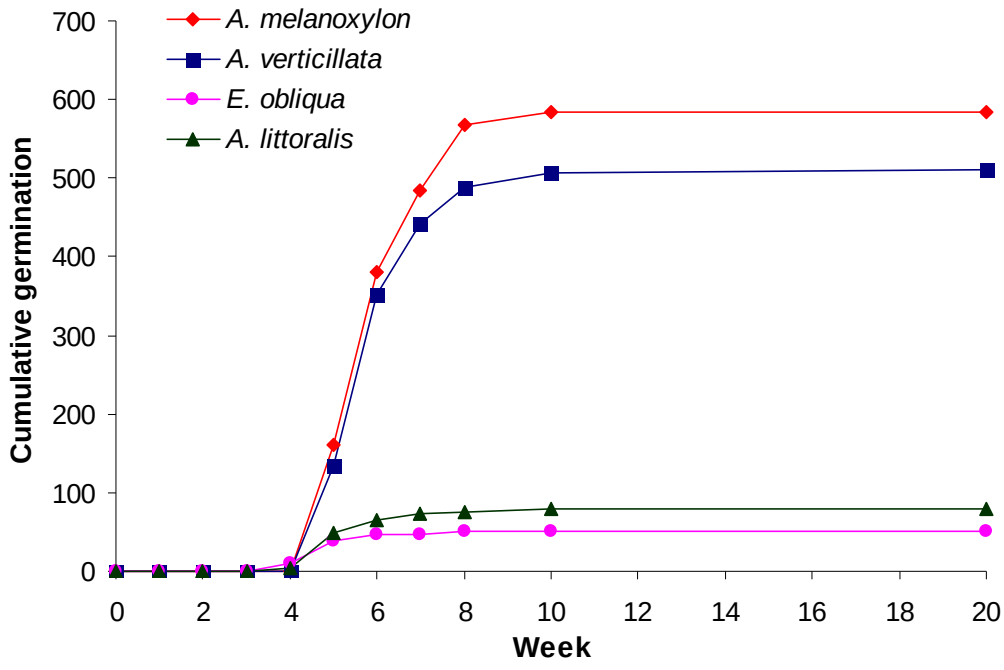


Figure 3.9: Total cumulative germination of seedlings over 20 weeks for each species used in the field experiment. Values are summed across all treatments (number of seeds sown for each species was 9000).

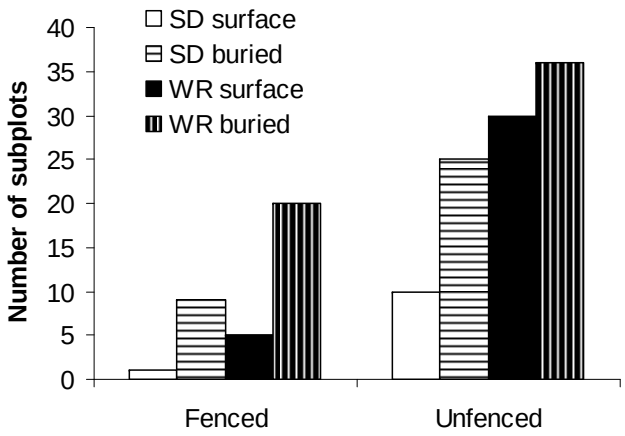
Interactive effects of experimental treatments

The four species sown did not show differences in emergence among blocks (Table 3.2) but the effect of fencing was very strong (at $p < 0.0005$) for each species, with germination numbers consistently lower in the fenced areas (Figure 3.10). The previous treatments of either fire or ground disturbance did not have a significant effect on the germination of the two *Acacia* species and *Allocasuarina*, but did produce a significant effect on the germination of *E. obliqua* within fenced areas only. However, *E. obliqua* only germinated in 8 subplots (of the 180 sown plots), 7 of which had a previous history of fire and 1 which had a previous history of ground disturbance.

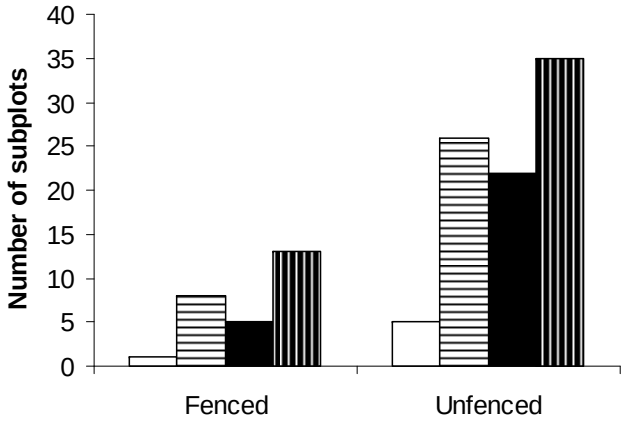
As no seedlings emerged in any of the control plots, a significant difference between the two ground preparation treatments and no ground preparation (control) was found for all species for both fenced and unfenced areas with the exception of *Allocasuarina* in the fenced area, which germinated in only 4 soil disturbance subplots and 7 weed removal subplots. The type of ground treatment also had a significant effect, in particular in the unfenced areas, in which all four species had their greatest emergence in subplots where weeds were removed from the entire surface to a depth of 5 cm (Figure 3.10). However, within the fenced treatment, germination of *A. melanoxylon* and *Allocasuarina* did not differ between soil disturbance and weed removal by scalping.

For both *Acacia* species, within each of the ground treatments in both the fenced and unfenced areas, germination was enhanced when seeds were buried rather than surface sown. For *Allocasuarina,* a significant effect of sowing treatment only occurred in interaction with the fenced and weed removal treatments. More seeds germinated when seeds were surface sown rather than buried. However, whether seeds were buried or surface sown did not have an effect on the germination of *E. obliqua* within either of the ground treatments.

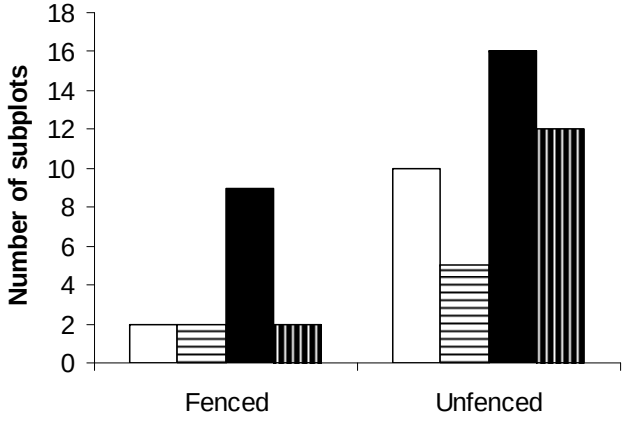
a. ***Acacia melanoxylon***



b. ***Acacia verticillata***



c. ***Allocasuarina littoralis***



d. ***Eucalyptus obliqua***

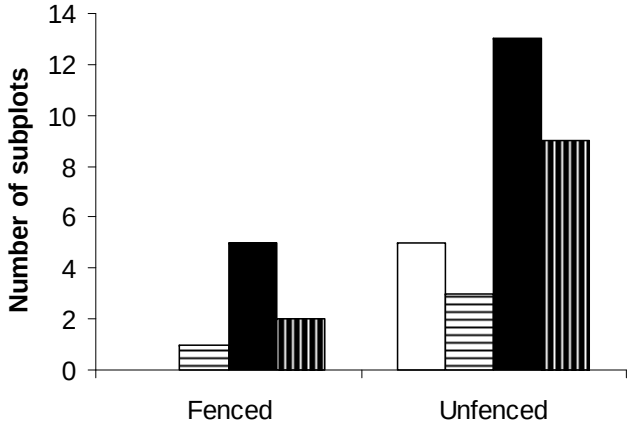
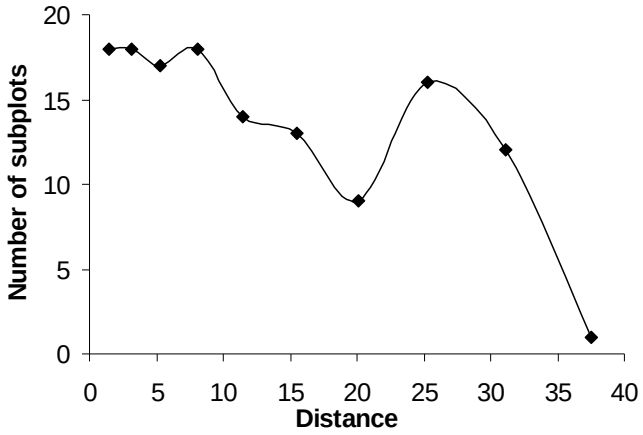
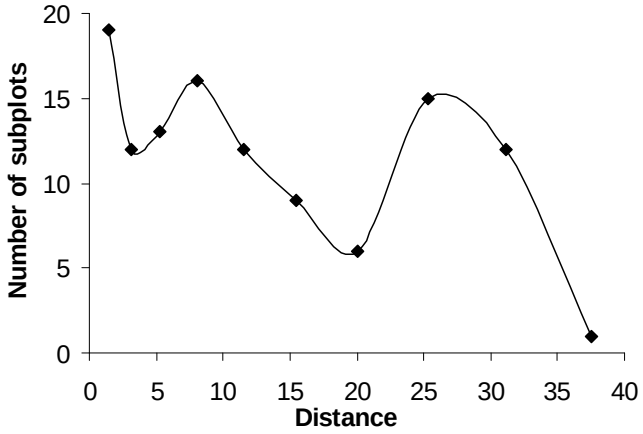


Figure 3.10: The cumulative number of subplots in which seedlings emerged after 10 weeks as influenced by fencing, ground preparation and sowing treatment. SD = soil disturbance; WR = weed removal.

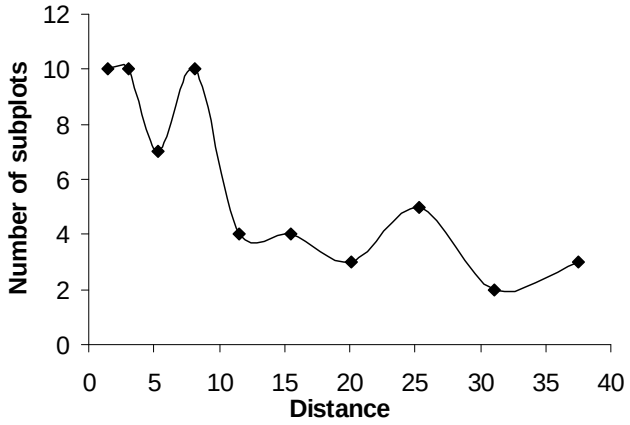
a. ***Acacia melanoxylon***



b. ***Acacia verticillata***



c. ***Allocasuarina littoralis***



d. ***Eucalyptus obliqua***

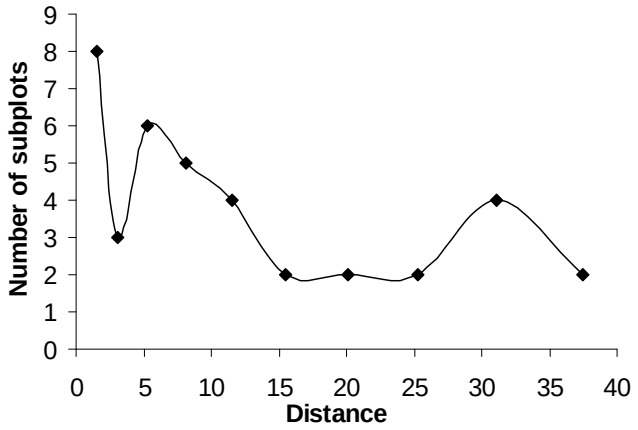


Figure 3.11: The cumulative number of subplots in which seedlings emerged after 10 weeks as influenced by distance from the native forest. SD = soil disturbance; WR = weed removal

Distance from the forest edge had a significant effect on the germination of both *Acacia* species and *Allocasuarina*. Germination of *Acacia* seeds was significantly reduced in plots at the greatest distance (37.5 m) from the remnant *Eucalyptus* forest, with the number of subplots in which seedlings emerged significantly lower at this distance in comparison to all other distances (Figure 3.11). *Allocasuarina* germination was adversely affected at distances of 11.5 m and greater from the remnant forest, with the number of *Allocasuarina* seedlings that emerged in rows 1 to 4 significantly greater than that which emerged in rows 5 to 10.

Table 3.2: Results of chi-square analysis on the cumulative number of plots in which seedling emergence occurred versus plots which recorded no germination over the range of experimental factors.

Acacia melanoxylon					
Factor		n	d.f.	Chi-Square	p-value
Block	N × S	720	1	2.94	0.087
Fencing	Fence (F) × Unfenced (UnF)	720	1	39.49	< 0.0005
Plot History	F Fire × F Ground Disturbance	320	1	0.03	0.858
	UnF Fire × UnF Ground Disturbance	320	1	0.13	0.718
Ground Preparation	F Soil Disturbance (SD) × F Weed Removal (WR) × Control	360	2	12.86	0.002
	F SD × F WR	320	1	7.22	0.007
	UnF SD × UnF WR × Control	360	2	32.43	< 0.0005
	UnF SD × UnF WR	320	1	13.90	< 0.0005
Sowing treatment	F SD buried × F SD surface × F SD no seed	160	2	24.32	< 0.0005
	F SD buried × F SD surface	80	1	7.31	0.007
	F WR buried × F WR surface × F WR no seed	160	2	50.96	< 0.0005
	F WR buried × F WR surface	80	1	13.09	< 0.0005
	UnF SD buried × UnF SD surface × UnF SD no seed	160	2	61.26	< 0.0005
	UnF SD buried × UnF SD surface	80	1	11.43	0.001
	UnFWRburied × UnF WRsurface × UnF WR no seed	160	2	114.20	< 0.0005
	UnF WR buried × UnF WR surface	80	1	3.12	0.077
Distance from forest	Row 1 × Row 2 × ……. × Row 10	320	9	33.04	< 0.0005
Acacia verticillata					
Block	N × S	720	1	3.74	0.053
Fencing	F × UnF	720	1	38.51	< 0.0005
Plot History	F Fire × F Ground Disturbance	320	1	0.36	0.546
	UnF Fire × UnF Ground Disturbance	320	1	0.06	0.802
Ground Preparation	F SD × F WR × Control	360	2	7.30	0.026
	F SD × F WR	320	1	3.28	0.070
	UnF SD × UnF WR× Control	360	2	26.00	< 0.0005
	UnF SD × UnF WR	320	1	10.60	0.001
Sowing treatment	F SD buried × F SD surface × F SD no seed	160	2	21.07	< 0.0005
	F SD buried × F SD surface	80	1	6.14	0.013
	F WR buried × F WR surface × F WR no seed	160	2	28.29	< 0.0005
	F WR buried × F WR surface	80	1	4.59	0.032
	UnF SD buried × UnF SD surface × UnF SD no seed	160	2	73.74	< 0.0005
	UnF SD buried × UnF SD surface	80	1	23.23	< 0.0005
	UnFWR buried × UnFWR surface × UnFWR no seed	160	2	97.76	< 0.0005
	UnF WR buried × UnF WR surface	80	1	10.31	0.001
Distance from forest	Row 1 × Row 2 × ……. × Row 10	320	9	33.37	< 0.0005

Allocasuarina littoralis					
Factor		n	d.f.	Chi-Square	p-value
Block	N × S	720	1	0.26	0.610
Fencing	F × UnF	720	1	14.70	< 0.0005
Plot History	F Fire × F Ground Disturbance	320	1	3.43	0.064
	UnF Fire × UnF Ground Disturbance	320	1	0.67	0.412
Ground Preparation	F SD × F WR × Control	360	2	5.79	0.055
	F SD × F WR	320	1	3.43	0.064
	UnF SD × UnF WR× Control	360	2	11.13	0.004
	UnF SD × UnF WR	320	1	4.54	0.033
Sowing treatment	F SD buried × F SD surface × F SD no seed	160	2	4.10	0.129
	F SD buried × F SD surface	80	1	0.0	1.0
	F WR buried × F WR surface × F WR no seed	160	2	21.38	< 0.0005
	F WR buried × F WR surface	80	1	5.17	0.023
	UnF SD buried × UnF SD surface × UnF SD no seed	160	2	20.23	< 0.0005
	UnF SD buried × UnF SD surface	80	1	2.05	0.152
	UnFWR buried × UnFWR surface × UnFWR no seed	160	2	35.33	< 0.0005
	UnF WR buried × UnF WR surface	80	1	0.88	0.348
Distance from forest	Row 1 × Row 2 × ……. × Row 10	320	9	19.29	0.023
Eucalyptus obliqua					
Block	N × S	720	1	0.45	0.505
Fencing	F × UnF	720	1	13.45	< 0.0005
Plot History	F Fire × F Ground Disturbance	320	1	4.62	0.032
	UnF Fire × UnF Ground Disturbance	320	1	0.15	0.701
Ground Preparation	F Fire SD × F Fire WR × Control (insufficient data)	200	1	n.a.	n.a.
	F GD SD × F GD WR × Control	200	2	10.88	0.004
	F GD SD × F GD WR	160	1	7.32	0.007
	UnF SD × UnF WR × Control	360	2	12.11	0.002
	UnF SD × UnF WR	320	1	7.21	0.007
Sowing treatment	F SD buried × F SD surface × F SD no seed	160	1	n.a.	n.a.
	F SD buried × F SD surface (Insufficient data)	80	1	n.a.	n.a.
	F WR buried × F WR surface × F WR no seed (GD plots only)	80	2	10.49	0.005
	F WR buried × F WR surface (GD plots only)	40	1	1.56	0.212
	UnF SD buried × UnF SD surface × UnF SD no seed	160	2	9.47	0.009
	UnF SD buried × UnF SD surface	80	1	0.56	0.456
	UnFWR buried × UnFWR surface × UnFWR no seed	160	2	27.19	< 0.0005
	UnF WR buried × UnF WR surface	80	1	1.00	0.317
Distance from forest	Row 1 × Row 2 × ……. × Row 10	320	9	11.23	0.260

Discussion

Viability and seed germination

The laboratory germination trials indicated that the purchased *Acacia* seed was highly viable whereas the locally sourced *Eucalyptus* and *Allocasuarina* seed was significantly less so, which was reflected by the seedling emergence results in the field trial. Although many scientific studies recommend using locally sourced seed from forest areas with similar soil type and rainfall as the restoration site (Handel *et al.* 1994; Koch and Samsa 2007), Broadhurst *et al*. (2008) found that local seed sourcing in many cases leads to poor restoration outcomes and suggests that sourcing should concentrate more on capturing high quality and genetically diverse seed.

The observed variation in viability and percentage germination could have also been affected by the maturity of the seed when collected and the depth of seed dormancy (Bell *et al.* 1995). The heat treatment applied to the *Acacia* seeds was shown to be effective in breaking seed dormancy by the high percentage germination (> 90%) achieved in the laboratory trials. Pre-treatments were not applied to the *E. obliqua* and *Allocasuarina* seeds based on past research (Boland *et al*. 1980; Langkamp 1987; Ralph 1994). However, germination of some tree and shrub species of eucalypt forest is known to be influenced by pre-treatment to remove dormancy (Bell 1994; Bell and Williams 1997). Further studies may be required to determine if there is seed dormancy in *E. obliqua* and *Allocasuarina littoralis*.

The laboratory trial was undertaken at 25 ^{o}C, which has been shown to be the optimal temperature for germination of *Allocasuarina* and *Acacia melanoxylon*, whereas the optimal temperature for *E. obliqua* germination is 15 ^{o}C (Boland *et al*. 1980; Langkamp 1987). Thus, the high temperature setting of the growth cabinet may have adversely affected the germination of the *E. obliqua* seed, though this does not explain the poor germination that occurred in the field trial. The poorer germination of *E. obliqua* seeds under the diurnal conditions reflects the findings of Facelli and Ladd (1996), who found that light suppressed germination of *E. obliqua* seeds by 67%.

Natural recruitment

The lack of native shrub or tree establishment on unsown plots indicates that the site has both low 'in situ resilience' and 'resilience by migration'. This lack of native reestablishment, even with the aid of gaps created in the grass sward by disturbances, is consistent with other studies that have found that restoration of improved pastures requires high levels of intervention to facilitate native restoration (Venning 1988; Toth *et al*. 1993; Yates and Hobbs 1997b; Walker *et al*. 2004; Slocum *et al.* 2006).

A study undertaken by Hellstrom *et al.* (2009) to determine whether the slow increase in species richness was explained by dispersal or micro-site limitations found that both factors restricted seedling establishment in impoverished semi-natural grasslands in northern Finland. Wassie *et al.* (2009) also found natural seed germination to be zero or low in open fields in northern Ethiopia and that soil scarification rather than removal from weed competition positively influenced germination.

Factors known to inhibit *Eucalyptus* regeneration in agricultural landscapes include cultivation, grazing by livestock and competition with exotic weeds, with natural regeneration depending on the coincidence of a proximal seed source, appropriate soil moisture and temperature, and an appropriate seed bed (Weinberg *et al*. 2011).

In their study on re-establishment of *Eucalyptus strezlecki*, Moxham and Dorrough (2008) found that weed competition limited regeneration. Exotic annuals have also been shown to limit the establishment of *Eucalyptus*, *Allocasuarina* and *Acacia* species in the West Australian wheatbelt (Prober and Smith 2009). As the removal of pasture vegetation in plots adjacent to the forest did not promote native seedling emergence, weed competition rather than distance from seed source also seems to be the limiting factor in the current study.

Many studies have identified poor soil conditions and soil moisture as important barriers to tree establishment (Battaglia and Reid 1993; Zimmerman *et al*. 2000; Skinner *et al.* 2009). The phosphorus (P) enriched soils of old fields seem to limit the establishment of

native plant species, with Standish *et al.* (2007) finding that the inability of native sclerophyll species to uptake phosphorus constrained their establishment on wheat old fields. Another study by Standish *et al*. (2008) found that P enrichment did not enhance the competitive ability of the exotic grass *Avena barbata*, thus concluding that the reduced native seed supply as well as the superior competitive abilities of invasive species both contributed to the persistence of exotics in old fields.

Phosphorus and nitrogen levels in the current study did not significantly vary with distance from forest edge which corresponds to the lack of natural recruitment across all distances. As barriers to natural regeneration did not vary with distance from native seed sources, weed competition and micro-site limitations seem to be the primary factors in limiting recovery. The outcomes from the direct seeding trials discussed below also indicate that site conditions were not favourable for natural recruitment.

Seedling emergence from direct seeding

The greater seedling emergence of both *Acacia* species in the field trial than *E. obliqua* and *Allocasuarina* reflects other findings where seed mass was a good predictor of germination success (Bell *et al.* 1995; Zimmerman *et al.* 2000; Geeves *et al*. 2008; Lindsay and Cunningham 2011). However the germinability of a broad spectrum of common woodland and forest species of northern New South Wales, with or without cues, has also been found to be negatively correlated with increasing seed size and larger growth forms (Clarke *et al*. 2000).

The larger seeded *Acacia* species had greater emergence success when buried, which concurs with the results of Doust *et al*. (2006), who found that larger seeded rainforest tree species had higher establishment rates on degraded tropical lands in northeast Queensland only where seeds were buried. While seed burial did not have an adverse effect on the emergence of *E. obliqua,* the greater emergence of *Allocasuarina* seedlings when surface sown indicates that small seeds are limited in terms of soil depth from which they can emerge relative to larger seeded species (Brown *et al*. 2003).

In their study on the emergence of shrubs and trees in grassy woodlands of the New England tablelands, Clarke and Davison (2001) found that eucalypt species emerged faster in the field than the shrub species and generally attained higher cumulative emergence than shrubs. In the present study, *E. obliqua* seedlings emerged 1 week prior to *Acacia* seedlings. However, the cumulative germination of *E. obliqua* was significantly lower than that of the *Acacia* species. In revegetation trials in the Tasmanian midlands, Close and Davidson (2003) found that direct seeded plots can result in a high ratio of acacia to eucalypt while Koch and Samsa (2007) found that establishment of jarrah (*Eucalyptus marginata*) from seed in restored sites is variable. The laboratory trial indicated the *E. obliqua* seed was less than 20 % viable whereas the *Acacia* seed had greater than 90 % viability. Thus, the amount of viable *E. obliqua* seed sown in the field trial was likely to have been much lower than the amount of viable *Acacia* seed.

In the month of September 2001, at the time of sowing, rainfall was below the long term average (78%). However, above average rainfall occurred in October 2001 (188%) which corresponded with the time that seedlings started to emerge for all four species. This above average rainfall continued for the next 3 months and thus does not explain why percentage emergence was lower in the field than the laboratory trials. Direct seeding success in the Tasmanian Midlands has been found to be greater in areas where mean annual precipitation exceeds 600 mm (Kirkpatrick *et al.* 2007) as is the case of the study site (780 mm). This pattern of lower percentages of seedling emergence under field conditions despite above average rainfall was also found by Clarke and Davison (2004), which they attributed to either mortality of germinable seed prior to emergence or a second dormancy effect.

Interactive effects of experimental factors

Protection from grazing by native and pest herbivores was found to adversely affect seedling emergence of all four sown species. The significantly less bare ground covering plots that had the combination of ground treatment and grazing indicates that

grazing was able to promote native seedling emergence through exotic vegetation control. This reflects the findings of Fensham and Kirkpatrick (1992), who found that protection from grazing on high nutrient sites can exclude native species.

Initial grazing has also been observed to reduce competition pressure from exotic annual pasture grasses on emerging natives of *Eucalyptus marginata* remnant woodlands by creating available sites for seed germination (Pettit and Froend 2001). Plots in the Western Australian wheatbelt which excluded grazing were observed to be dominated by annual exotic species (Hobbs 1989). Established shrubs in subalpine zones of Australia were also found to only spread onto areas where competition from herbs was reduced by grazing and unoccupied by snowgrass (Wimbush and Costin 1979).

However, Stohlgren *et al.* (1999) observed that exclosure studies looking at how grazing affects native species richness can produce inconsistent and questionable results because most exclosures are smaller than 1 ha and are thus compromised by edge effects. In particular, their investigation of plant diversity in Rocky Mountain grasslands found that 1 m^2 subplots in grazed sites had significantly higher native species richness than ungrazed plots whereas 1000 m^2 exclosures had nearly identical native or exotic species richness to grazed plots.

In the current study the effect of grazing alone was not sufficient to facilitate native seedling emergence in the seeded plots which did not have any ground preparation. Other studies have likewise found that degraded, invaded and isolated vegetation should not be managed solely by a single restoration technique such as grazing by native or introduced herbivores but should be integrated with other methods of vegetation control, such as weeding (Venning 1988; Lunt 1990; Yates and Hobbs 1997b; Kirkpatrick *et al*. 2007; Milchunas *et al.* 2011). Onan and Parsons (1980) found that conditions for mallee seedling establishment on cleared land in south-eastern Australia included low levels of rabbit grazing and minimal competition. In prealpine calcareous grasslands successful restoration was possible with the combination of grazing regimes and shrub clearing, but not possible with only shrub clearing (Barbaro *et al.* 2001). In these cases, grazing was

beneficial in providing suitable micro-sites for seedling germination by increasing the gap frequency within closed swards and reducing the competitiveness of exotic palatable species.

Of the two ground preparation treatments used in the field trial, the removal of pasture vegetation by scalping in combination with grazing produced the greatest seedling emergence for all sown species. Weed removal resulted in significantly less bare ground cover 10 weeks after application than in plots which had soil disturbance, indicating that removal from competition with weeds was critical for successful seedling emergence. Evidence from many restoration studies have found that weed control significantly improves the success of direct seeding of woody sclerophyll species in exotic grasslands (Knight *et al.* 1998; Yates *et al*. 2000; Kirkpatrick *et al*. 2007; Moxham and Dorrough 2008).

Soil nutrient levels did not significantly vary in relation to distance from the native remnant forest. However there was a significant increase in soil moisture from the direction of the forest to the pasture as influenced by the terrain at the field site. Although young seedlings are susceptible to surface drying on exposed soils during the critical months following germination (Skinner *et al.* 2009) and watering in combination with weed control was found to result in more successful seedling emergence from direct seeding trials in the Tasmanian Midlands (Kirkpatrick *et al.* 2007), lack of soil moisture was unlikely to be a factor in the present study due to the greater than average rainfall that occurred during the field trial. Also, as bare ground cover was not affected by distance from forest, it would seem that the lower *Acacia* and *Allocasuarina* emergence that occurred at greater distances from the forest could have been influenced by damping off fungi encouraged by high levels of soil moisture. A low tolerance of *Acacia* species to waterlogging in the early stages was also observed by Gonzalez-Munoz *et al*. (2011).

Seed burial in combination with ground preparation was found to enhance *Acacia* seedling emergence, though under laboratory conditions, *A. verticillata* had greater

germination in the diurnal conditions whereas *A. melanoxylon* had the same percentage germination in diurnal and dark conditions. *E. obliqua* emergence in the field was not affected by seed burial which is in contrast to the results of the laboratory trial and previous studies which have found that *E. obliqua* germination was substantially lower under light than dark conditions (Bell *et al*. 1995; Facelli and Ladd 1996).

Conclusions

The results from the field trial highlight that restoration of native vegetation in improved pastures requires the reintroduction of native seed to facilitate seedling emergence. The absence of natural recruitment, even with the aid of gaps created in the grass sward by disturbances, indicates that a reduced native seed supply in combination with the superior competitive abilities of exotic grasses are limiting factors to native reestablishment. Under the conditions of the field trial, direct seeding proved most successful in drier plots where weed control was prolonged by the combination of ground preparation and grazing by native and pest herbivores. Direct seeding of *Acacia* was significantly more successful than that of *E. obliqua* and *Allocasuarina*, which both had < 1% cumulative seedling emergence, indicating that the site conditions were not favourable for the reestablishment of *E. obliqua* and *Allocasuarina* from seed.

Chapter Four

The response of re-introduced native seedlings to variation of pasture community and weed control techniques

Introduction

Where site conditions are not favourable for natural recruitment or establishment from seed, as found in the previous chapter, seedling re-introductions are likely to be more successful in overcoming the competitive thresholds that limit recruitment from seed as they are less dependent on the climatic and abiotic conditions (Cooper and McDonald 2000; Hobbs and Cramer 2007; Pueyo *et al*. 2009; Prober *et al*. 2008).

Although planting tube stock is generally considered more expensive than direct seeding or allowing for natural recruitment, in highly modified areas where the recovery of natural vegetation without intervention is limited, revegetation may be a cheaper strategy in the long term (Sieben and Reinecke 2008).

The successful reintroduction of native tree species on pastoral land often depends on zones of decreased competition for water, nutrients and root space (Fensham and Kirkpatrick 1992; Hitchmough *et al*. 1996; Parrotta *et al*. 1997; Yates and Hobbs 1997; Appleby 1998). There have been numerous studies into the effects of cultivation and weed control on both conifers and eucalypts. However the wide range of treatments, soil and site conditions confuses the interpretations of these results (Holz *et al*. 1999). Grantz *et al.* (1998) found that in highly variable arid and semi-arid environments that site preparation does not guarantee successful revegetation. In temperate environments, reducing root competition has been recommended (McConnaughay and Bazzaz 1990; Zacharek 1996; Morgan 1997), which can be achieved by removing the top 5-10 cm of soil and the live root mass of competing species by scalping.

Where exotic weed cover is abundant, site preparation that involves scalping has been reported to promote seedling growth and to be an effective treatment for re-establishing conifers in boreal woods (Lieffers *et al*. 1993; Man *et al*. 2009); *Eucalyptus* and *Acacia* in central west New South Wales (Geeves *et al*. 2008); and fir and pine in south-western Oregon (Flint and Childs 1987; Schneider *et al*. 1998). It has also been reported to be more effective than herbicide treatments at reducing the live ground cover of exotic weeds for as much as 3 years following treatment (Geeves *et al*. 2008).

However, from their survey of revegetation sites in the Tasmanian midlands Close and Davidson (2003) recommend the repeated applications of knockdown herbicide prior to planting of tube stock along with post planting weed controls and Turvey (1996) found that the chemical weed control of *Imperata* grassland increased the total volume production of *Eucalyptus urophylla* 11-fold.

There may, however. be situations where site preparation is not always practical (Cogliastro *et al*. 1997) and since weed control costs may be amongst the highest in the establishment of tree seedlings (Little and Rolando 2001), seedling growth could be optimised by selecting species that suit the site conditions. Improved seedling performance can result from the careful selection of planting locations based on drainage and vegetation type (Crowell and Lane 2001; Raulings *et al.* 2007). Poorly-drained sites in particular must be revegetated with trees with waterlogging adaptations (Bell 1999). Thus, it is important to identify the competitive abilities and natural distribution of the individual species which are to be reintroduced.

Cooper and McDonald (2000) found that the establishment rates of transplanted seedlings on mined peatlands were highly influenced by hydrological conditions, with *Carex* species having the greatest survival in wetter sites while *Kobresia* species established on drier sites, and recommended determining the optimal conditions of transplanted species to increase survivability.

The reconstruction of the native vegetation of the field site prior to agricultural disturbance identified that *E. ovata* woodland occurred in the low-lying *Juncus* spp. community, *E. obliqua* open-forest occurred on the higher ground dominated by the *Senecio* spp. community and *E. obliqua/E. amygdalina* open-forest on the steeper northern slopes dominated by bracken. This variation in microtopography and drainage conditions across the site is likely to affect the survival and growth of *Eucalyptus* seedlings (Battaglia and Reid 1993). To determine if the natural distributions of the *Eucalyptus* species present in the remnant vegetation influences survival and growth; seedlings of *E. amygdalina, E. obliqua, E. ovata, E. viminalis* and *Allocasuarina littoralis* were planted in the three weed communities present at the site. As the duration of weed control has been found to be dependent on site conditions (Adams *et al*. 2003) the study also assessed the effectiveness of weed control in promoting seedling growth in the three different weed communities.

Thus the questions asked in this chapter are:

1. Does site selection based on a species natural distribution optimise seedling survival and growth?
2. Which weed control treatment is most effective for promoting seedling growth?
3. Is there a significant difference between sites in the effectiveness of weed control treatments?

Materials and methods

Study area and ground preparation

The three exotic weed communities chosen for this experiment were dominated respectively by *Juncus* spp. (rush), *Senecio* spp. (fireweed) and *Pteridium esculentum* (bracken) communities (Chapter 2). The weed control treatments selected were a scalping treatment, an herbicide treatment and a combination of scalping and above ground removal of weeds 9 months after scalping. To protect the seedlings from grazing, fences which excluded wallabies, rabbits and possums were constructed. The construction of the three fences took place in July and September 2000. The sizes of the fences were 23 m by 30 m to contain all the experimental treatments. Within each fenced area 10 replications of each weed control treatment were randomly applied to 3 m^2 quadrats. Thus, including the control quadrats, there were 40 plots in each area.

For the herbicide treatment, the plots were brush cut in October 2000. The herbicide was applied a week later, followed by a second application after two weeks due to the amount of live plant material still prevalent in the plots. The herbicide used was Glyphosate CT Broad acre herbicide, active constituent 450 g/L glyphosate. The herbicide was applied at a rate of 10 ml/L water, with 2 L on average used for every plot. The scalping treatment was applied in November 2000 using a mattock to remove the vegetation and top 3-5 cm of soil. In April 2001 the weed control treatments were re-applied. However, during the autumn application the herbicide was only applied once with 1L of herbicide at the above mentioned concentration used on each plot. The third weed control treatment involved ground vegetation being removed from the plots, using a brush cutter, on the 22 January 2002, 9 months after the scalping occurred.

Plate 4.1: The Bracken fernland pasture community and the remnant *E. obliqua/E. amygdalina* forest. Photograph taken November 1999.

Plate 4.2: In the foreground is the *Juncus* spp. rushland, mid-way up the slope is the *Senecio* spp. herbfield and on the highest ground is the *E. obliqua* remnant forest. Photograph taken November 1999.

Plate 4.3: The first application of weed control treatments, herbicide (bottom left) and scalping (centre right), that were applied to the *Juncus* spp. rushland. Photograph taken April 2001.

Recording of vegetation cover in quadrats

The projective foliage cover values of each taxon in all quadrats were recorded before the application of weed control treatments (October 2000) and 2 years later. The cover values of all individual species present were assessed by visual estimation using the Braun-Blanquet cover abundance scale. The midpoints of the cover classes were then used in analysis. For some species, identification to species level at that time of year could not be made confidently and the species were aggregated to generic level. These genera included *Juncus*, *Poa* and *Senecio*. The cover of bare ground and litter was recorded on the two above occasions as well as at 6, 9 and 12 months after planting (November 2001, February 2002 and May 2002).

Plant material and treatments

The seeds of four *Eucalyptus* and one *Allocasuarina* species were collected in May and June 2000 from within approximately 10 km from the field site at Saltwater River in south-eastern Tasmania. In July 2000 seeds were placed on a seed raising mix of sand, peat and vermiculite (1:1:1) in seedling trays placed on a heating mat set at 25 degrees Celsius (ºC). Once seedlings had cotyledon leaves they were pricked out into tubes which were 50 mm × 50 mm square and 120 mm deep. The potting mix was composed of composted pine bark, sand, peat and A1 horizon loamy sand soil collected in the remnant forest at the study site in the respective ratio of 6:4:1:2 with the potting mix having a pH range between 6.5 –7. The site soil was used in an attempt to introduce mycorrhizal fungi that may assist in the growth of the seedlings. Slow release Osmocote was added to the potting mix at a concentration of 300 g for every 50 L of potting mix.

After three weeks all germinants were planted into tubes and placed in the glasshouse at the temperature range of 10 – 20 ºC (and hand watered daily). Tubes were arranged into trays that held 96 tubes thus 24 of each species were placed in each tray and the trays were then randomly arranged in the glasshouse (randomised block design). On the 23 October 2000 seedlings were transferred from the glasshouse to a shade house with a mild coastal climate until they were planted at the study site in Autumn 2001.

Seedling treatments

The planting treatments consisted of planting a seedling of each species in each quadrat in a regular pattern (at least one metre from the edge of the quadrat and other seedlings). The position of the seedling in the pattern was randomly allocated each time to reduce any effect of position within the quadrats on seedling growth.

Holes approximately 50 mm × 50 mm square and 125 mm deep were dug for the seedlings using a hand and foot operated Hamilton Tree planter. Seedlings were first soaked in water tubs before planting and then three days later they were watered with approximately 500 ml of water. This was the only artificial watering the plants received

after planting. The height of seedlings was recorded on planting with increases in height and seedling survival recorded every three months after planting for two years. Stakes were also put in place to support and protect seedlings from being blown over in windy conditions and were removed once seedlings had outgrown the stakes. All stakes were removed by the end of the two year experimental period.

Soil properties

The volumetric water content of each quadrat was measured on four occasions, August 2001, March 2001, June 2002 and September 2002, using a Time Domain Reflectometry (TDR) soil moisture probe. Prior to measuring in the field, the TDR probe was calibrated using soil collected from the site. Soil samples were also collected from the three locations in December 2002. Ten samples were randomly collected from non-treatment areas within each community to assess the soil nutrient status of the three locations. The soil cores were 10 cm deep and 5 cm in diameter.

The soil samples were air-dried and then coarsely sieved to remove roots, litter and stones. The following chemical analyses were completed:

- available phosphorus using Bray extractable method (Jackson 1958);
- total phosphorus using the Perchloric/Nitric/Sulphuric acid digestion method (Allen 1974);
- total nitrogen using the Kjeldahl method (Jackson 1958);
- total organic carbon using the Walkley and Black method (Rayment and Higginson 1992).

The composite of the soil sampling units from the same location was then divided into four samples on which the following analyses were also carried out:

- pH using the 1:5 soil/water suspension (Rayment and Higginson 1992);
- electrical conductivity using the 1:5 soil/water extract (Rayment and Higginson 1992).

Analysis

Soil properties

The effect of weed community on soil moisture and chemistry was analysed using the Kruskal-Wallis test. Where a significant difference was detected by the Kruskal-Wallis test, the Mann-Whitney U test was used to test differences between two samples.

Vegetation cover

The cover values of the species and site attributes were entered into DECODA (Minchin 1991), and total species cover as well as native and exotic species richness was determined for each quadrat. The percentage frequency occurrence of each species within each treatment before and after the application of the weed control treatments was also determined.

Comparisons were made of the effects of the treatments within each community on the change in absolute native cover; relative native cover; native richness; absolute exotic cover; relative exotic cover; exotic richness; native/exotic cover ratio; dominant native genera cover; exotic grass cover; exotic herb cover; and, bare ground within each of the three weed communities. Absolute cover is defined as the actual cover that was recorded using the Braun-Blanquet cover abundance scale whereas relative cover is defined as the absolute cover divided by the total vegetation cover of the plot. Thus, the total relative cover of all plants adds up to 100 percent.

The normality of the data was tested using plots of the standardised residuals versus fitted values and normal probability plots. It was necessary to log transform some of the data, for example log10 (%cover + 0.5) was used. Mean change in cover data from 2000 and 2002 was analysed using one-way ANOVA (or the Kruskal-Wallis test for non-parametric data) at the significance level $p < 0.05$. When a one-way ANOVA or Kruskal-Wallis test gave a significant result, pairwise comparisons were made using Fishers pairwise analysis (individual error rate = 0.05) or Mann-Whitney U test respectively.

Species richness was calculated as the number of recorded taxa per quadrat. The mean change in species richness from 2000 and 2002 was calculated for both native and exotic species within each weed community. The change in exotic richness between the treatments was analysed using one-way ANOVA. However native richness data were insufficient to carry out a parametric analysis of means thus the change in native richness was compared using Kruskal-Wallis test.

Species frequency was calculated as the number of plots that the species/species groups were recorded in within each weed community. The mean change in frequency was calculated for all exotic species; exotic grasses; all native species; native grasses; native herbs and any individual species which had notable changes in frequency. Fishers exact probability test was used to analyse the changes in frequency between each treatment type and the control.

Seedling survival and growth

The effect of location and weed treatment on the survival of each of the five species was analysed at 3, 9 and 24 months using Fishers exact probability test. For each of the three weed communities a General Linear Model (GLM) analysis was undertaken to assess the significance of treatment and species identity and their combined effect on the Relative Growth Rate (RGR) of seedlings after 24 months. RGR = (Height at 24 months – Initial height)/Initial height. Within the GLM analysis, pairwise comparisons were undertaken using the Tukey Honest Significance Difference (HSD) test at the 95% confidence level. The normality of the data was tested using plots of the standardised residuals versus fitted values and normal probability plots. To meet the assumptions of normality it was necessary to transform some of the data. Square root (RGR + 0.5) or log (RGR + 0.5) was used.

The effects of weed community, weed treatment and their combined effect on each species was also analysed using the GLM analysis, with pairwise comparisons undertaken using the Tukey HSD test at the 95% confidence level. Due to the poor survival of *A. littoralis, E. amygdalina* and *E. obliqua* in the *Juncus* community, only

data from the *Senecio* and Bracken communities was used for these three species. Where the GLM analysis produced a significant combined effect, one-way ANOVAs were undertaken to test the significance of treatment within each weed community on the mean RGR of the five species. If there was a significant result from the one-way ANOVA, Fishers pairwise analysis at the 95% confidence level was used to determine significant differences between pairs (Sokal and Rohlf 1995; Dytham 1999).

Results

Soil properties

The soils of all three weed communities were acidic, although the level of acidity varied significantly from community to community. Soil from the Bracken and *Senecio* communities also had similar total organic carbon and total nitrogen levels which varied significantly from the *Juncus* community (Figure 4.1). However, both the *Senecio* and *Juncus* communities had a level of available phosphorus that was significantly higher than that recorded for the Bracken community (Figure 4.2).

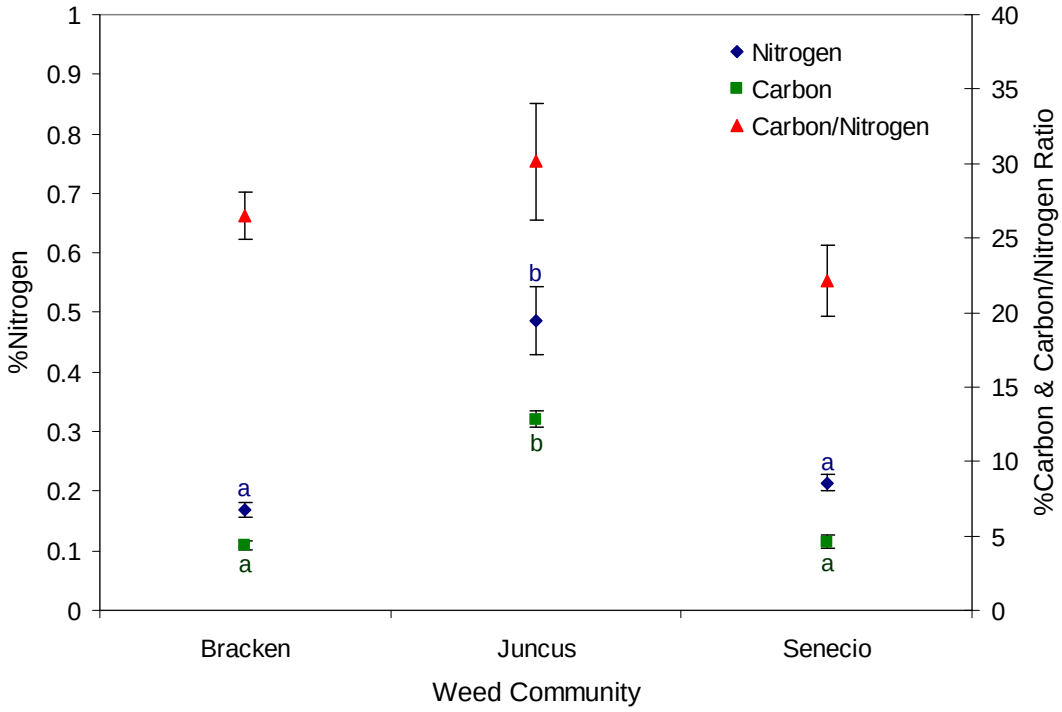


Figure 4.1: Mean nutrient levels in the soil of the three weed communities; Bracken, *Juncus* and *Senecio*. (Error bars = standard error). Note: two y-axes used. %Nitrogen data points are read using the left hand y-axis whereas the %Carbon and C/N data points are read using the right hand y-axis. Means with no letters in common are significantly different at $p < 0.05$ (Mann-Whitney U test).

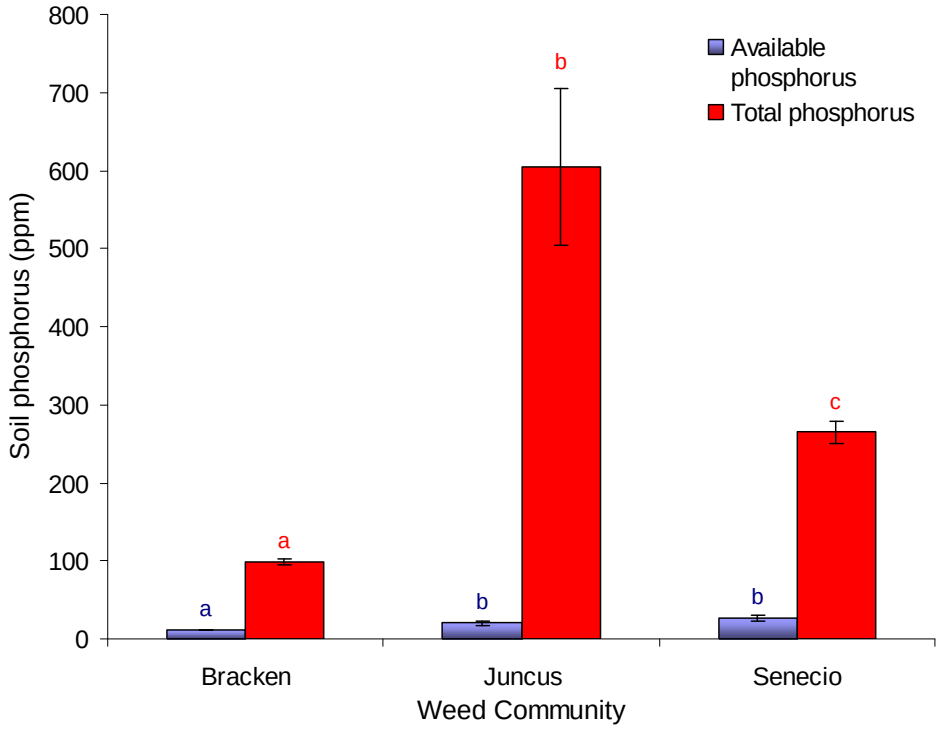


Figure 4.2: Mean total and available phosphorus levels in three weed communities. (Error bars = standard error). Means with no letters in common are significantly different at $p < 0.05$ (Mann-Whitney U test).

At each of the four times measurements were taken the soil moisture of each weed community varied significantly (Figure 4.3). The *Juncus* community had the highest soil moisture in each season whereas the Bracken community was the driest at each recording. The *Juncus* community also experienced seasonal waterlogging with standing surface water common during winter and spring. The lowest soil moisture for all weed communities was recorded in March 2002, which reflects the rainfall data obtained by the Bureau of Meteorology (Figure 4.4).

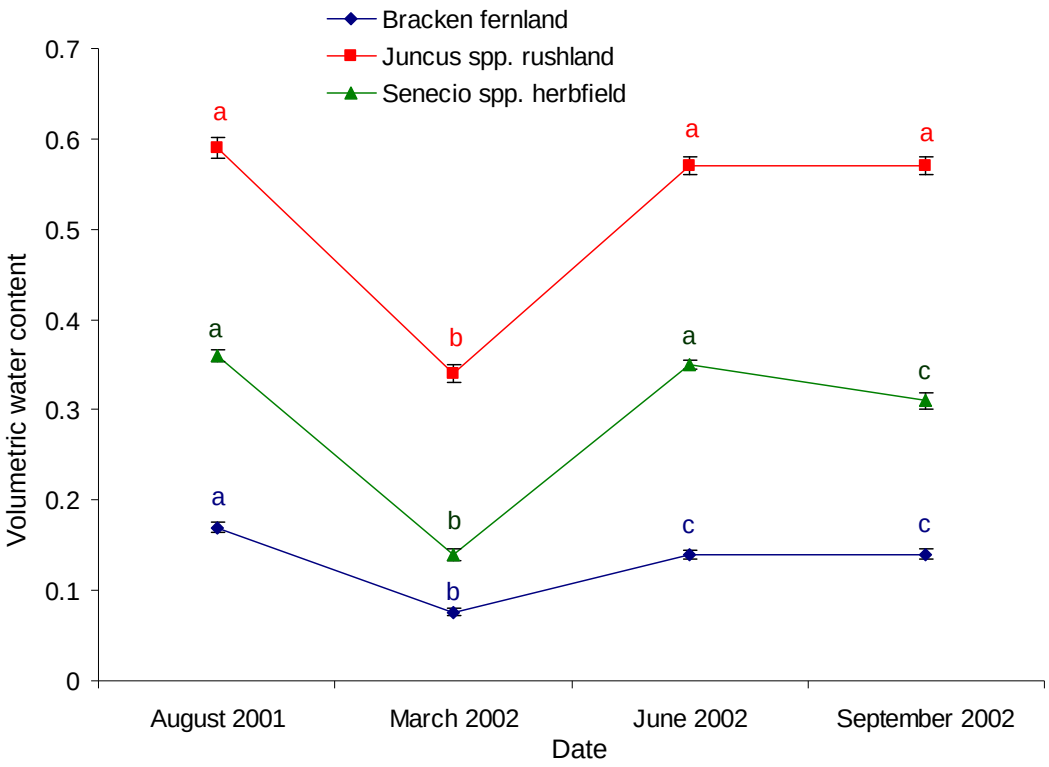


Figure 4.3: The mean volumetric water content for each of the three weed communities taken four times over the course of a year. (Error bars = standard error). Means with no letters in common are significantly different at $p < 0.05$ (Mann-Whitney U test).

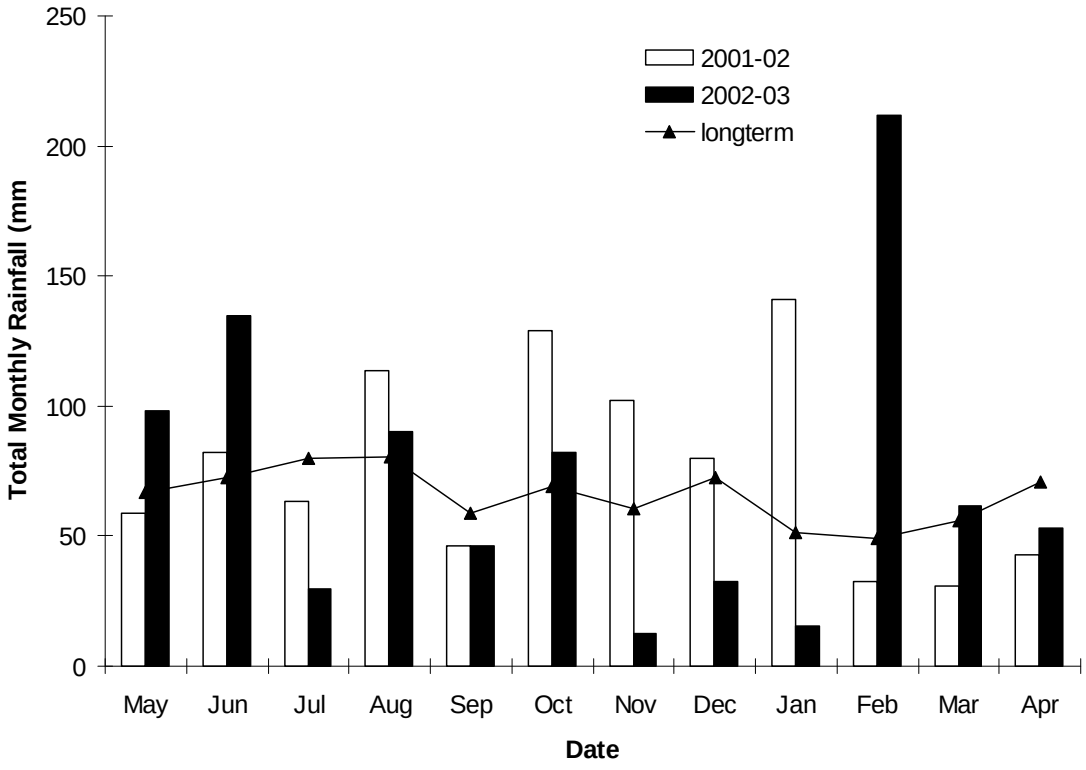


Figure 4.4: The monthly rainfall for Saltwater River from May 2001 until April 2003 and the long term monthly average (Bureau of Meteorology).

The effect of weed control treatments on vegetation composition

Juncus community

i. Plant cover

Within the *Juncus* location, exotic cover decreased after two years in all treatments without any significant difference between the treatments. This pattern was reversed for the native species which increased in cover after two years with again no significant difference between treatments. However, a significant difference in *Juncus* spp., exotic herb, exotic grass and bare ground cover occurred between treatments (Table 4.1).

The herbicide treatment was the only treatment to have a negative impact on *Juncus* spp. cover (-5.44 ± 3.00 %) which was significantly different from the increased *Juncus* spp. cover in the scalp (15.33 ± 5.16 %), scalp & slash, (Sc/Sl), (10.94 ± 6.51 %) and control plots (19.50 ± 5.64 %) (Table 4.1). Exotic grass cover decreased in all plots with a significant decrease of -64.00 ± 8.27 % occurring in the plots treated by Sc/Sl, whereas exotic herb cover had an increase of 20.70 ± 10.8 % in the Sc/Sl plots as opposed to the decrease of its cover in the other three treatments. Bare ground cover was significantly higher in the scalp plots than the herbicide and control plots (Table 4.1).

Table 4.1: The effect of experimental treatments on the vegetation attributes of the plots. a. The change in percentage cover values assessed by visual estimation using the Braun-Blanquet cover abundance scale. b. The change in species richness. All figures are mean ± standard error; n = 9. p-values indicate results of one-way ANOVA or Kruskal-Wallis test. Where a one-way ANOVA or Kruskal-Wallis test gave a significant result, superscripts indicate the results of pairwise comparisons made using Fishers pairwise analysis (individual error rate = 0.05) or Mann-Whitney U test respectively. Treatments with different letters are significantly different at $p < 0.05$.

Juncus location	Herbicide	Control	Scalp	Scalp + Slashing	p-value, F statistic
a. Plant cover					
Absolute native cover	15.78 ± 6.26[a]	20.22 ± 5.70[a]	25.61 ± 7.01[a]	35.61 ± 9.13[a]	0.252, 1.43
Relative native cover	15.98 ± 4.80 [a]	16.08 ± 4.16 [a]	20.48 ± 4.23 [a]	22.55 ± 3.87 [a]	0.628, 0.59
Absolute exotic cover	-56.72 ± 9.64[a]	-47.94 ± 6.57[a]	-60.11 ± 7.29[a]	-43.3 ± 15.10[a]	0.602, 0.63
Relative exotic cover	-15.35 ± 5.22[a]	-16.15 ± 4.10[a]	-21.20 ± 4.18[a]	-23.27 ± 3.92[a]	0.519, 0.77
Native/ Exotic Ratio	0.27 ± 0.09[a]	0.23 ± 0.06[a]	0.33 ± 0.10[a]	0.37 ± 0.08[a]	0.635, H=1.71, df=3
Juncus spp. cover	-5.44 ± 3.00[a]	19.50 ± 5.64[b]	15.33 ± 5.16[b]	10.94 ± 6.51[b]	0.011, 4.35
Exotic grass cover	-38.61 ± 5.62[a]	-30.94 ± 5.55[a]	-43.78 ± 7.46[a]	-64.00 ± 8.27[b]	0.012, 4.28
Exotic herb cover	-18.61 ± 5.76[a]	-17.39 ± 3.36[a]	-16.33 ± 6.69[a]	20.7 ± 10.80[b]	0.010, H=11.32, df=3
Bare ground	-1.44 ± 1.63[a]	0.06 ± 0.06[a]	1.22 ± 0.45[b]	1.56 ± 0.46[b]	0.004, H=15.65, df=3
b. Species richness					
Native richness	1.67 ± 0.29[a]	-0.44 ± 0.24[b]	2.67 ± 0.37[a]	2.44 ± 0.38[a]	<0.0005, H=22.01, df=3
Exotic richness	-1.89 ± 0.46[a]	-1.33 ± 0.41[a]	0.44 ± 0.34[b]	1.56 ± 0.58[b]	<0.0005, 12.28 df=3

ii. Species richness

The plots that were treated by scalping and Sc/Sl had a significantly higher exotic richness, 0.44 ± 0.34 and 1.56 ± 0.58 respectively, than the untreated (-1.33 ± 0.41) and herbicide (-1.89 ± 0.46) plots. Both scalping treatments increased in exotic richness after two years whereas the herbicide treated and control plots had a decrease in exotic richness. All three weed treatments of herbicide, scalping and Sc/Sl had a positive effect on native richness (1.67 ± 0.29, 2.67 ± 0.37, and 2.44 ± 0.38 respectively) which was a significant difference to the negative response of native richness in the control plots (-0.44 ± 0.24) (Table 4.1).

Though the one-way ANOVA analysis of the average increase in native richness showed no significant differences between the treatments (Table 4.1), an analysis of native richness using Fishers exact probability test showed that overall there was an increase in total (pooled across replicates) native richness across all treatments and the control plots, with the scalp treatment significantly promoting native richness. Six native species that were not present prior to treatment application occurred in the scalp plots, which was significantly higher than the appearance of 1 extra species found in the control plots. Both the herbicide and Sc/Sl treatments had an increase of 3 and 2 native species after two years (Table 4.2).

Table 4.2: The effect of experimental treatments on species richness. The figures indicate the increase in species richness that occurred within each treatment (using data pooled across replicates). Superscripts indicate results of Fishers exact probability test. Treatments with different letters are significantly different at $p < 0.05$.

Species richness	Herbicide	Control	Scalp	Scalp + Slashing
Native	3^{ab}	1^{a}	6^{b}	2^{ab}
Exotic	2^{a}	1^{a}	4^{a}	5^{a}

After two years the number of exotic species present increased in all treatments and the control, with the scalp and Sc/Sl treatments having the highest increase of 4 and 5 respectively. This positive effect of the scalp and Sc/Sl treatments on exotic richness corresponds with the positive mean increase of exotic richness in these two treatments

(Table 4.1). The lower increases of 2 and 1 exotic species in the herbicide and control were not significantly different from the scalp and Sc/Sl treatments (Table 4.2) differing in that respect from the mean exotic richness results which were significantly different (Table 4.1).

iii. Frequency

The Fishers exact probability test indicated that an increase in native herb frequency occurred within the treated plots. The scalp and Sc/Sl treatments had native herbs occurring in 5 and 6 more plots while the largest increase of 7 occurred in the herbicide treatment which was significantly higher than the increase of 2 within the control plots (Table 4.3).

Carex appressa was initially recorded in one control and one scalp plot but was not observed in the second survey. *Juncus planifolius* and *Rumex brownii* were two native species that were present in all treatments at the second survey but were not recorded in the initial one. *Isolepis* spp. was also not recorded in any plots at the start of the trial but was observed in the weed treated plots two years after treatments were applied (Table 4.3), with its frequency significantly higher in the scalp and Sc/Sl plots than in the control and herbicide plots. *Gonocarpus teucrioides*, *Hypericum japonicum* and *Juncus bufonius* were the three native species that were only present in the scalp plots post treatment application though each one of these species was observed only once.

The exotic grass *Agrostis* spp. was also not recorded in the initial vegetation survey but was present in all treatments and the control in the survey conducted two years after the treatment applications with its frequency significantly higher in the control and scalp plots (Table 4.3). *Epilobium* spp. was another exotic species only observed after treatment application, it was recorded in only 1 scalp and Sc/Sl plot each, 2 control plots and in 6 herbicide plots which was significantly higher than the scalp and Sc/Sl recording. The thistle *Cirsium vulgare* was present in 7 control plots initially but was not recorded in any control plots after two years and its frequency was reduced from 6 to 3 in the scalp plots. However, in the herbicide plots its frequency increased from 4 to 5

and it remained in six of the Sc/Sl plots, both results being significantly different to the control. *Lolium perenne* was recorded in all plots initially but decreased in all treatments and significantly so in the herbicide plots where it was recorded in only 1 plot. *Trifolium repens* was another exotic species which was present in all plots initially but its frequency was significantly decreased in the herbicide and control plots (-7 and -8 respectively), whereas it continued to be present in all scalp and Sc/Sl plots (Table 4.3).

Table 4.3: The effect of experimental treatments on the plot frequency of vegetation attributes. All figures are the change in the number of quadrats in which the attributes were present: n = 9. Superscripts indicate results of Fishers exact probability test. Treatments with different letters are significantly different at $p < 0.05$.

Species	Herbicide	Control	Scalp	Scalp + Slashing
All exotics	0 [a]	0 [a]	0 [a]	0 [a]
Exotic grasses	0 [a]	0 [a]	0 [a]	0 [a]
Exotic herbs	-3 [ab]	-5 [b]	0 [a]	0 [a]
All natives	0 [a]	-1 [a]	0 [a]	0 [a]
Native grasses	-2 [a]	-1 [a]	0 [a]	0 [a]
Native herbs	7 [a]	2 [b]	5 [ab]	6 [ab]
Juncus spp.	-2 [a]	-1 [a]	0 [a]	0 [a]
Isolepis spp.	2 [a]	0 [a]	7 [b]	7 [b]
Senecio spp.	7 [b]	1 [a]	3 [ab]	2 [a]
Agrostis spp.	2 [a]	9 [b]	8 [b]	4 [a]
Epilobium sp.	6 [b]	2 [ab]	1 [a]	1 [a]
Cirsium vulgare	1 [a]	-7 [b]	-3 [ab]	0 [a]
Lolium perenne	-8 [b]	-2 [a]	-2 [a]	-4 [a]
Trifolium repens	-7 [a]	-8 [a]	0 [b]	0 [b]

Bracken community

i. Plant cover

Within the Bracken community native plant cover increased in the control (40.72 ± 6.06 %), herbicide (18.11 ± 6.42 %), and scalp (31.44 ± 5.29 %) plots after two years which was significantly different to the decrease of -15.83 ± 3.64 % that occurred in Sc/Sl plots (Table 4.4). Exotic plant cover also increased in the control plot (47.00 ± 7.32 %) which was significantly different to the effect the three treatments had on exotic cover. In the herbicide (20.50 ± 6.90 %) and scalp (7.39 ± 9.25 %) plots exotic cover did increase after two years but in a significantly less proportion than in the control plots indicating that the treatments had a negative effect on exotic plant growth. The most significant response in comparison to the control was the reduction in exotic growth that was recorded in the Sc/Sl plots (-11.67 ± 8.37 %), however this was not statistically different to the response recorded in the scalp plots.

After two years there was an increase in the native/exotic ratio in control plots which also occurred in the herbicide and scalp treatments whereas the Sc/Sl treatment had a significant decrease of -0.166 ± 0.133 in the ratio of natives to exotics thus proving to have a marginal negative effect on the native species cover.

As *Pteridium esculentum* was the only native species to be observed in the initial survey the change in the cover of *Pteridium esculentum* very closely resembled the changes recorded by total native cover, with the negative effect that occurred in the Sc/Sl plots significantly different to the increases in *Pteridium esculentum* that occurred in the control and other two treatments. The changes in exotic grass cover also mimicked the results of the total exotic cover with each of the three treatments suppressing the growth of exotic grass cover in comparison to the control. However, the treatments had an insignificant effect on exotic herb cover. After two years the two scalping treatments were able to sustain a significant bare ground cover with bare ground persisting on average over a quarter of the plot when treated by Sc/Sl (Table 4.4).

Table 4.4: The effect of experimental treatments on the vegetation attributes of the plots. a. The change in percentage cover values assessed by visual estimation using the Braun-Blanquet cover abundance scale. b. The change in species richness. All figures are mean ± standard error; n = 9. p-values indicate results of one-way ANOVA or Kruskal-Wallis test. Where a one-way ANOVA or Kruskal-Wallis test gave a significant result, superscripts indicate the results of pairwise comparisons made using Fishers pairwise analysis (individual error rate = 0.05) or Mann-Whitney U test respectively. Treatments with different letters are significantly different at $p < 0.05$

Bracken location	Herbicide	Control	Scalp	Scalp + Slashing	p-value, F statistic
a. Plant cover					
Absolute native cover	18.11 ± 6.42 [a]	40.72 ± 6.06 [a]	31.44 ± 5.29 [a]	-15.83 ± 3.64 [b]	<0.0005, 15.82
Relative native cover	1.60 ± 4.14 [ab]	10.20 ±3.70 [ac]	13.33 ± 3.44 [c]	-6.34 ± 4.30 [b]	0.005, 5.15
Absolute exotic cover	20.50 ± 6.90 [a]	47.00 ± 7.32 [b]	7.39 ± 9.25 [ac]	-11.67 ± 8.37 [c]	<0.0005, 9.46
Relative exotic cover	-1.60 ± 4.14 [ab]	-10.20 ± 3.70 [ac]	-13.33 ± 3.44 [c]	6.34 ± 4.30 [b]	0.005, 5.15
Native/Exotic Ratio	0.02 ± 0.11 [ab]	0.24 ± 0.09 [ac]	0.35 ± 0.08 [c]	-0.17 ± 0.13 [b]	0.007, 4.89
Bracken spp. cover	16.67 ± 5.59 [a]	40.00 ± 5.91 [a]	30.00 ± 5.32 [a]	-17.78 ± 3.37 [b]	<0.0005, H= 21.16, df=3
Exotic grass cover	17.28 ± 6.09 [a]	48.39 ± 6.01 [b]	12.17 ± 7.39 [a]	-11.28 ± 4.25 [c]	<0.0005, 16.51
Exotic herb cover	3.22 ± 4.38 [a]	-1.39 ± 3.38 [a]	-4.78 ± 3.16 [a]	-0.39 ± 5.60 [a]	0.619, 0.60
Bare ground	0.78 ± 0.28 [a]	1.06 ± 0.37 [a]	5.39 ± 1.84 [b]	25.28 ± 5.67 [c]	<0.0005, H= 27.45, df=3
b. Species richness					
Native richness	0.67 ± 0.44 [a]	0.33 ± 0.24 [a]	0.67 ± 0.33 [a]	1.11 ± 0.26 [a]	0.219, H=5.49 df=3
Exotic richness	-0.11 ± 0.35 [a]	0.00 ± 0.41 [a]	-1.22 ± 0.22 [b]	-2.00 ± 0.37 [b]	0.003, H=15.23, df=3

ii. Species richness

Before the application of treatments the only native species recorded in the experimental plots was *Pteridium esculentum*, two years later the average native richness increased in all plots without any significant difference recorded between the treatments and control. An analysis of the change in total number of native species observed within each treatment (Table 4.5) also shows an insignificant difference between the control and weed control treatments.

Nine exotic species were observed in the Bracken location prior to treatment application and a significant reduction in average exotic species richness was observed both in the scalp (-1.22 ± 0.222) and Sc/Sl (-2.00 ± 0.373) treatments (Table 4.4). However, an analysis of the pooled exotic richness of each treatment does not show a significant difference between the control and treatments (Table 4.6).

Table 4.5: The effect of experimental treatments on species richness. The figures indicate the change in species richness that occurred within each treatment (using data pooled across replicates). Superscripts indicate results of Fishers exact probability test. Treatments with different letters are significantly different at $p < 0.05$.

Species richness	Herbicide	Control	Scalp	Scalp + Slashing
Native (w/o *Pteridium esculentum*)	5 [a]	3 [a]	4 [a]	4 [a]
Exotic	2 [a]	1 [a]	-1 [a]	-1 [a]

iii. Frequency

As *Pteridium esculentum* was the dominant species in all plots it was not included in the analysis of the frequency of native species in experimental plots so it could be determined whether the frequency of native species that were absent prior to treatment application were influenced by the different treatments. Unlike the analysis on the change in mean native species richness (Table 4.4) and the pooled native richness (Table 4.5) which showed no significant differences between treatments, the frequency analysis showed that the increase in frequency of plots with native species was significantly

higher in the Sc/Sl treatment in comparison to the control. Native species were found in 7 of the 9 Sc/Sl plots compared to the 2 control plots that had recordings of native species. This trend continued for the native herb data but not for native grasses indicating that the Sc/Sl treatment had a positive effect on the emergence of native herb species (Table 4.6). *Gonocarpus teucriodes* in particular was observed in 6 Sc/Sl plots and in 3 scalp plots but not in any control plots (Table 4.6).

Table 4.6: The effect of experimental treatments on the plot frequency of vegetation attributes. All figures are the change in the number of quadrats in which the attributes were present: n = 9. Superscripts indicate results of Fishers exact probability test. Treatments with different letters are significantly different at $p < 0.05$.

Species	Herbicide	Control	Scalp	Scalp + Slashing
All exotics	0^{a}	0^{a}	0^{a}	0^{a}
Exotic grasses	0^{a}	0^{a}	0^{a}	0^{a}
Exotic herbs	0^{a}	0^{a}	0^{a}	0^{a}
Dactylis glomerata	0^{a}	1^{a}	-5^{b}	-3^{ab}
Holcus lanatus	-1^{ab}	0^{a}	0^{a}	-4^{b}
All natives (w/o bracken)	3^{ab}	2^{a}	4^{ab}	7^{b}
Native grasses	3^{a}	1^{a}	0^{a}	1^{a}
Native herbs	1^{a}	1^{a}	4^{ab}	7^{b}
Bracken	0^{a}	0^{a}	0^{a}	0^{a}
Aotus ericoides	1^{a}	0^{a}	0^{a}	2^{a}
Chiloglottis spp.	2^{a}	1^{a}	0^{a}	1^{a}
Epacris impressa	0^{a}	0^{a}	1^{a}	0^{a}
Eucalyptus obliqua	1^{a}	0^{a}	1^{a}	0^{a}
Gonocarpus micranthus	1^{a}	0^{a}	1^{a}	1^{a}
G. teucrioides	0^{a}	0^{a}	3^{ab}	6^{b}
Luzula flaccida	1^{a}	1^{a}	0^{a}	0^{a}
Monotoca glauca	0^{a}	1^{a}	0^{a}	0^{a}

Senecio community

i. Plant cover

Two years after the application of weed control treatments, native cover lessened in the treated plots but increased in the control. However, these results were not statistically significant (Table 4.7). The reverse trend was recorded for exotic cover but the results were also not statistically significant. A significant result was observed in the ratio of native to exotics which decreased significantly in the both the scalp (-0.41 ± 0.22 %) and Sc/Sl (-0.40 ± 0.08 %) treatments while an increase was noted in the control plots (0.166 ± 0.14 %).

Senecio spp. was the dominant native species group and thus the changes in *Senecio* cover were closely aligned to the native cover results with the exception that the statistical analysis determined that the Sc/Sl (-29.72 ± 5.68 %) treatment had a significant adverse effect on *Senecio* cover in relation to the control (5.89 ± 9.86 %). As native herb cover was mostly composed of *Senecio* spp. its results also followed a similar trend but the statistical analysis showed that both the scalp and Sc/Sl treatments (-12.61 ± 6.42 % and -32.00 ± 7.08 %respectively) had a significant adverse affect on native herb cover (Table 4.7).

Table 4.7: The effect of experimental treatments on the vegetation attributes of the plots. a. The change in percentage cover values assessed by visual estimation using the Braun-Blanquet cover abundance scale. b. The change in species richness. All figures are mean ± standard error; n = 9. p-values indicate results of one-way ANOVA or Kruskal-Wallis test. Where a one-way ANOVA or Kruskal-Wallis test gave a significant result, superscripts indicate the results of pairwise comparisons made using Fishers pairwise analysis (individual error rate = 0.05) or Mann-Whitney U test respectively. Treatments with different letters are significantly different at $p < 0.05$

Senecio location	Herbicide	Control	Scalp	Scalp + Slashing	p-value , F statistic
a. Plant cover					
Absolute native cover	-2.90 ± 10.60[a]	5.83 ± 9.94 [a]	-11.94 ± 6.35 [a]	-31.61 ± 6.97 [a]	0.057, H=7.52, df=3
Relative native cover	0.44 ± 8.44 [ab]	7.03 ± 7.18 [a]	-17.49 ± 8.28 [bc]	-24.17 ± 3.78 [c]	0.013, 4.20
Absolute exotic cover	2.10 ± 14.40[a]	-17.40 ± 10.70[a]	27.20 ± 20.10[a]	29.60 ± 13.50[a]	0.126, 2.05
Relative exotic cover	-0.44 ± 8.44 [ab]	-7.03 ± 7.18 [a]	17.49 ± 8.28 [bc]	24.17 ± 3.78 [c]	0.013, 4.20
Native/ Exotic Ratio	-0.00 ± 0.16 [ab]	0.17 ± 0.14 [a]	-0.41 ± 0.22 [b]	-0.40 ± 0.08 [b]	0.031, 3.34
Senecio spp. cover	-1.70 ± 10.30[a]	5.89 ± 9.86 [a]	-11.50 ± 6.29 [a]	-29.72 ± 5.68 [b]	<0.0005, 7.95
Exotic grass cover	2.10 ± 16.40 [a]	-25.20 ± 12.50[a]	23.20 ± 17.80[a]	13.40 ± 12.00[a]	0.098, 2.29
Exotic herb cover	-0.44 ± 4.91 [a]	7.78 ± 4.21 [a]	3.94 ± 6.37 [a]	16.22 ± 9.46 [a]	0.482, H=2.46, df=3
Native herb cover	-3.60 ± 11.00[ab]	5.83 ± 9.94 [a]	-12.61 ± 6.42 [b]	-32.00 ± 7.08[c]	0.001, 7.67
Bare ground	0.17 ± 0.08[a]	0.89 ± 0.41[a]	0.33 ± 0.08[a]	0.22 ± 0.09[a]	0.279, H=3.84 df=3
b. Species richness					
Native richness	-0.78 ± 0.28[a]	-0.67 ± 0.24[a]	-0.89 ± 0.20[a]	-0.67 ± 0.24[a]	0.887, H=0.64 df=3
Exotic richness	-1.78 ± 0.66[a]	-2.11 ± 0.66[a]	-0.56 ± 0.63[a]	-0.89 ± 0.61[a]	0.288, 1.31

ii. Species richness

The mean native species richness decreased in all plots two years after treatment application with no significant differences between treatments and the control. This also occurred with the mean exotic richness which decreased in all three treatments and the control, the results of which were again statistically similar (Table 4.7). The analysis of the pooled richness data (Table 4.8) revealed that no change in species richness occurred in the herbicide, scalp and Sc/Sl treatments while a decrease of only 1 native species occurred in the control plots. This loss was *Acaena novae-zelandiae* which was recorded in one control plot initially.

Table 4.8: The effect of experimental treatments on species richness. The figures indicate the change in species richness that occurred within each treatment (using data pooled across replicates). Superscripts indicate results of Fishers exact probability test. Treatments with different letters are significantly different.

Species richness	Herbicide	Control	Scalp	Scalp + Slashing
Native	0 [a]	-1 [a]	0 [a]	0 [a]
Exotic	-1 [a]	-1 [a]	2 [a]	1 [a]

iii. Frequency

A summary of the change in frequency of the major vegetation attributes is given in Table 4.9. From the table it can be seen that there were not any statistical differences in the frequency of either native or exotic species groups. Unlike the mean cover analysis in which the scalp and Sc/Sl treatments were shown to have an adverse effect on mean *Senecio* cover the frequency analysis shows that the weed treatments had little impact on the plot frequency of *Senecio* spp. The frequency analysis also indicated a drop in the plot frequency of two native species, *Acaena novae-zelandiae* and *Hydrocotyle hirta*, which was consistent across all treatments and thus not treatment induced. Also of note is the occurrence of *Juncus* species in the treatment plots which were not observed in the initial survey. However, the frequency of occurrence was too small to be of significance.

Table 4.9: The effect of experimental treatments on the plot frequency of vegetation attributes. All figures are the change in the number of quadrats in which the attributes were present: n = 9. Superscripts indicate results of Fishers exact probability test. Treatments with different letters are significantly different at $p < 0.05$.

Species	Herbicide	Control	Scalp	Scalp + Slashing
All exotics	0^{a}	0^{a}	0^{a}	0^{a}
Exotic grasses	0^{a}	0^{a}	0^{a}	0^{a}
Exotic herbs	0^{a}	0^{a}	0^{a}	0^{a}
All natives	0^{a}	0^{a}	0^{a}	-1^{a}
Native reeds	1^{a}	0^{a}	2^{a}	2^{a}
Native herb	0^{a}	0^{a}	0^{a}	-1^{a}
Senecio spp.	0^{a}	0^{a}	0^{a}	-1^{a}
Acaena novae-zelandiae	-3^{a}	-1^{a}	-5^{a}	-1^{a}
Hydrocotyle hirta	-6^{a}	-5^{a}	-5^{a}	-6^{a}
Juncus pallidus	1^{a}	0^{a}	0^{a}	0^{a}
Juncus pauciflorus	1^{a}	0^{a}	1^{a}	0^{a}
Juncus planifolius	0^{a}	0^{a}	1^{a}	2^{a}

Seedling survival and growth

Seedling survival

After the first three months since planting there were no significant differences in seedling survival between the treatments for all species with the exception of *Eucalyptus obliqua*. Only half the seedlings of this species survived in the herbicide treatment within the *Juncus* community which was significantly different from its 100% survival in the Bracken and *Senecio* locations. However within the *Juncus* location itself there were no significant differences amongst the treatments (Table 4.10).

Table 4.10: The seedling survival rates of five species planted in three plant communities under different weed control treatments 3 months after planting. Eight individuals of *Eucalyptus amygdalina* and *E. obliqua* were planted per treatment while nine individuals were planted for the other three species. Within each species, superscripts indicate results of Fishers exact probability test. Treatments with different letters are significantly different at $p < 0.05$. The descriptions of the treatments are, C = control, H = herbicide, S = scalp and Sc/Sl = Scalp + Slash.

Location	Treatment	*Allocasuarina littoralis*	*Eucalyptus amygdalina*	*E. obliqua*	*E. ovata*	*E. viminalis*
Bracken	C	9^{A}	8^{A}	8^{A}	9^{A}	9^{A}
	H	9^{A}	8^{A}	8^{A}	9^{A}	9^{A}
	S	9^{A}	8^{A}	8^{A}	9^{A}	9^{A}
	Sc/Sl	9^{A}	8^{A}	8^{A}	9^{A}	9^{A}
Juncus	C	8^{A}	7^{A}	6^{AB}	9^{A}	9^{A}
	H	9^{A}	7^{A}	4^{B}	9^{A}	9^{A}
	S	9^{A}	7^{A}	7^{AB}	9^{A}	9^{A}
	Sc/Sl	9^{A}	8^{A}	6^{AB}	9^{A}	9^{A}
Senecio	C	9^{A}	8^{A}	8^{A}	9^{A}	9^{A}
	H	9^{A}	8^{A}	8^{A}	9^{A}	9^{A}
	S	9^{A}	8^{A}	8^{A}	9^{A}	9^{A}
	Sc/Sl	9^{A}	8^{A}	8^{A}	9^{A}	9^{A}

After nine months, once again there were not any significant differences in survival across the treatments in the Bracken and *Senecio* locations for all five species but seedling survival was adversely affected by treatments within the *Juncus* location for *Allocasuarina littoralis*, *Eucalyptus amygdalina*, *E. obliqua* and to a lesser extent for *E.*

viminalis. *Allocasuarina littoralis*, *Eucalyptus amygdalina* and *E. obliqua* had a significant decrease in seedling survival within the *Juncus* location irrespective of treatment type; indicating that location rather than weed control treatment had a stronger impact on survival. However, the survival of *E. viminalis* was only adversely affected when planted in the scalp treatment within the *Juncus* location (Table 4.11).

Table 4.11: The seedling survival rates of five species planted in three plant communities under different weed control treatments 9 months after planting. Eight individuals of *Eucalyptus amygdalina* and *E. obliqua* were planted per treatment while nine individuals were planted for the other three species. Within each species, superscripts indicate results of Fishers exact probability test. Treatments with different letters are significantly different at $p < 0.05$. The descriptions of the treatments are, C = control, H = herbicide, S = scalp and Sc/Sl = Scalp + Slash.

Location	Treatment	*Allocasuarina littoralis*	*Eucalyptus amygdalina*	*E. obliqua*	*E. ovata*	*E. viminalis*
Bracken	C	9^{A}	7^{A}	8^{A}	9^{A}	9^{A}
	H	9^{A}	6^{AB}	7^{A}	9^{A}	9^{A}
	S	9^{A}	7^{A}	8^{A}	9^{A}	9^{A}
	Sc/Sl	9^{A}	8^{A}	8^{A}	9^{A}	9^{A}
Juncus	C	3^{CD}	2^{BC}	1^{C}	9^{A}	9^{A}
	H	5^{BCD}	1^{C}	1^{C}	9^{A}	9^{A}
	S	3^{CD}	2^{BC}	0^{C}	9^{A}	5^{B}
	Sc/Sl	2^{D}	0^{C}	2^{BC}	9^{A}	8^{AB}
Senecio	C	7^{ABC}	6^{AB}	8^{A}	9^{A}	9^{A}
	H	9^{A}	6^{AB}	7^{A}	9^{A}	9^{A}
	S	9^{A}	7^{A}	6^{AB}	9^{A}	9^{A}
	Sc/Sl	8^{AB}	8^{A}	8^{A}	9^{A}	8^{AB}

The survival trends that were established after nine months did not differ greatly from the survival that was observed after two years (Table 4.12). All five species still had high survival in all treatments within the Bracken and *Senecio* locations. The environment within the *Juncus* location once again was shown to have an adverse affect on *Allocasuarina littoralis*, *Eucalyptus amygdalina* and *E. obliqua* regardless of the treatment. As with the results obtained after nine months *E. viminalis* had a significantly lower survival rate when planted in the scalp treatment within the *Juncus* location.

Table 4.12: The seedling survival rates of five species planted in three plant communities under different weed control treatments 2 years after planting. Eight individuals of *Eucalyptus amygdalina* and *E. obliqua* were planted per treatment while nine individuals were planted for the other three species. Within each species, superscripts indicate results of Fishers exact probability test. Treatments with different letters are significantly different at $p < 0.05$. The descriptions of the treatments are, C = control, H = herbicide, S = scalp and Sc/Sl = Scalp + Slash.

Location	Treatment	*Allocasuarina littoralis*	*Eucalyptus amygdalina*	*E. obliqua*	*E. ovata*	*E. viminalis*
Bracken	C	9^{A}	7^{A}	7^{A}	9^{A}	9^{A}
	H	8^{AB}	5^{ABC}	6^{A}	9^{A}	8^{AB}
	S	9^{A}	7^{A}	8^{A}	8^{A}	8^{AB}
	Sc/Sl	9^{A}	8^{A}	8^{A}	9^{A}	9^{A}
Juncus	C	1^{CD}	2^{BCD}	1^{B}	9^{A}	9^{A}
	H	5^{BC}	0^{D}	1^{B}	9^{A}	9^{A}
	S	1^{CD}	1^{CD}	0^{B}	9^{A}	5^{B}
	Sc/Sl	0^{D}	0^{D}	0^{B}	9^{A}	7^{AB}
Senecio	C	6^{AB}	6^{AB}	7^{A}	9^{A}	9^{A}
	H	8^{AB}	5^{ABC}	7^{A}	9^{A}	9^{A}
	S	9^{A}	7^{A}	6^{A}	9^{A}	9^{A}
	Sc/Sl	8^{AB}	7^{A}	8^{A}	9^{A}	8^{AB}

Seedling growth - Effects of weed treatment and species type on seedling growth within three weed communities.

Bracken community

Within the Bracken community there was not a significant combined effect of treatment and species type (GLM, $F = 1.24$, $p = 0.261$) on the mean relative growth rate (RGR) of seedlings, though treatment and species type independently had significant effects (GLM, $F = 4.41$, $p = 0.005$ and $F = 5.50$, $p < 0.0005$ respectively) on mean RGR. These results indicate that the five tree species responded similarly to the ground preparation treatments, with Figure 4.5 showing that *E. ovata* was the only species not to have the greatest growth in the control plots.

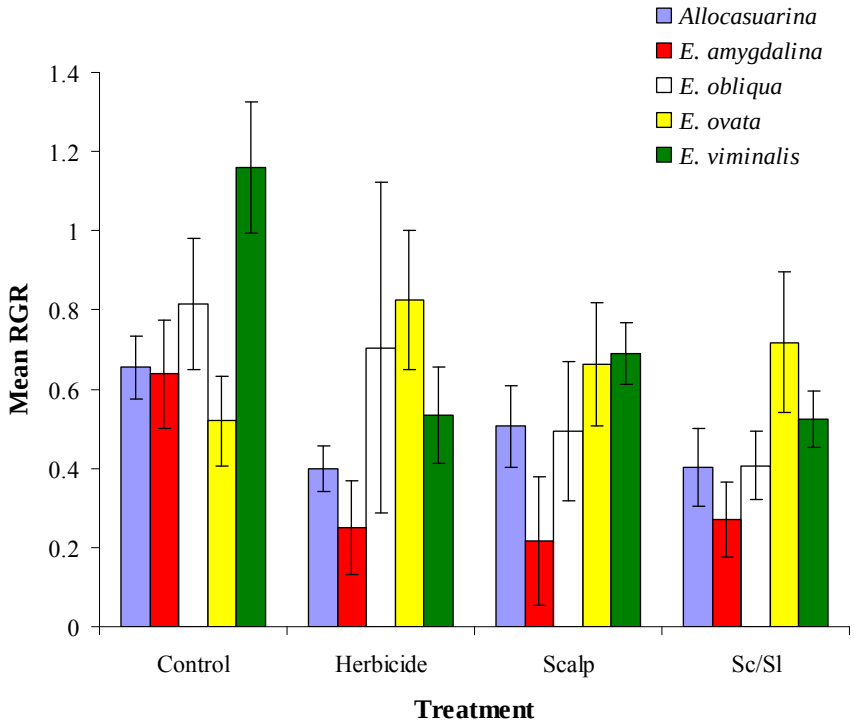


Figure 4.5: The mean Relative Growth Rate (RGR) of five tree species grown under different weed control treatments in the Bracken community. Columns represent the mean ± standard error.

When pooling species data across treatments, *E. amygdalina* was found to have the poorest overall growth (0.35 ± 0.07 RGR) which was significantly lower than both *E. viminalis* (0.73 ± 0.07) and *E. ovata* (0.68 ± 0.08) growth (Figure 4.6).

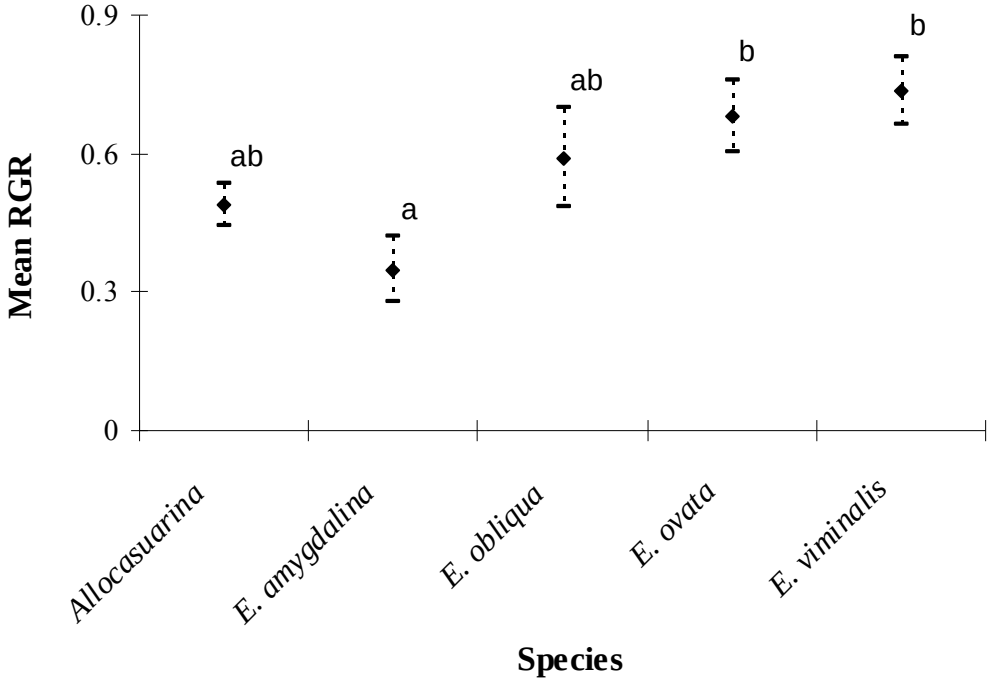


Figure 4.6: The response of five tree species planted in the Bracken community. Data points show the mean RGR ± standard error. Means with no letters in common are significantly different at $p < 0.05$ (Tukey HSD test).

The Tukey HSD comparison of treatment effects showed that seedling RGR was significantly greater in the control plots (0.76 ± 0.07) than the plots that had weed control treatments applied, with no significant differences found between the treated plots (Figure 4.7).

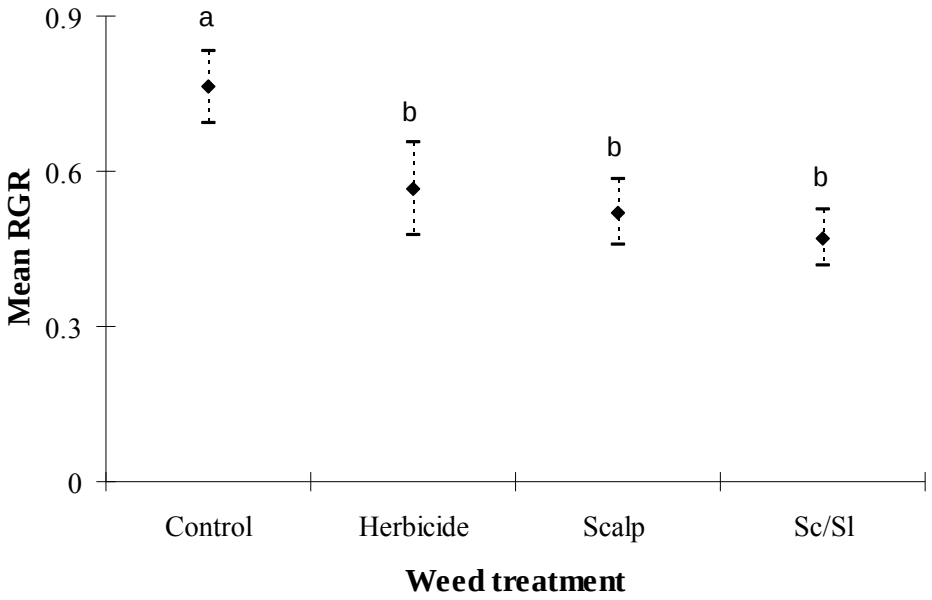


Figure 4.7: The effect of weed treatment on the combined RGR pooled of five tree species planted in the Bracken weed community. Data points show the mean ± standard error. Means with no letters in common are significantly different at $p < 0.05$ (Tukey HSD test).

Juncus community

Due to the high mortality of *Allocasuarina littoralis, E. amygdalina* and *E. obliqua* in this location the growth analysis was only carried out on *E. ovata* and *E. viminalis* data. As within the Bracken community the combined effect of treatment and species type was not significant (GLM, $F = 2.25$, $p = 0.092$) whereas the individual effects of treatment and species were significant (GLM, $F = 17.58$, $p < 0.0005$ and $F = 10.08$, $p = 0.002$ respectively). So once again the results indicate that the growth of these two *Eucalyptus* species were similarly affected by the treatments (Figure 4.8).

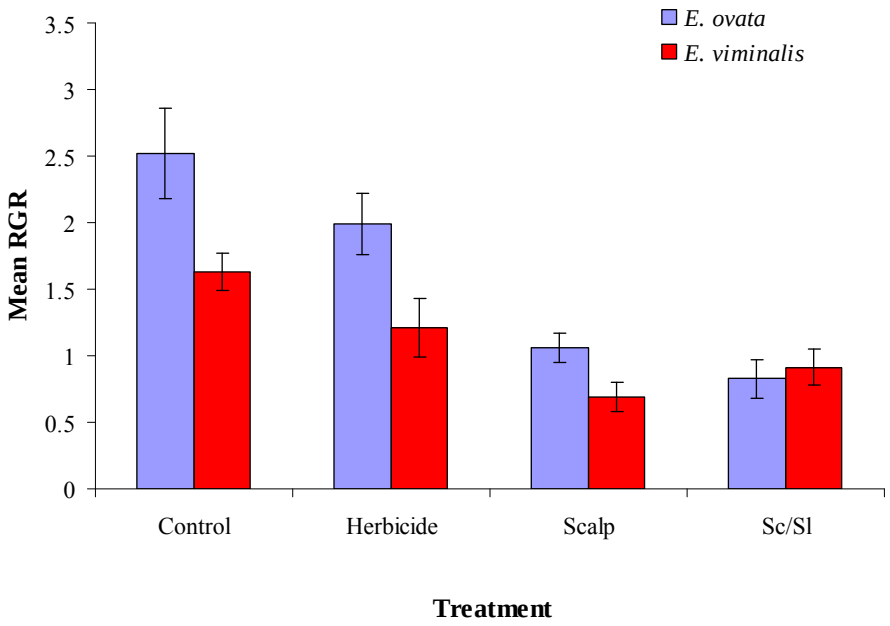


Figure 4.8: The mean Relative Growth Rate (RGR) of *E. ovata* and *E. viminalis* when grown under different weed control treatments in the *Juncus* weed community. Columns represent the mean ± standard error.

The mean growth of *E. ovata* (1.60 ± 0.16 RGR) was found to be significantly greater than *E. viminalis* (1.18 ± 0.10 RGR) as shown in Figure 4.9. In regards to treatment effects, the greatest growth occurred in the control plots (2.08 ± 0.21 RGR) which was significantly different to seedling growth within the treated plots (Figure 4.10). The herbicide treatment did have a significantly better effect on seedling growth than the scalp and Sc/Sl treatments though it was still significantly less than the growth recorded in the control plots.

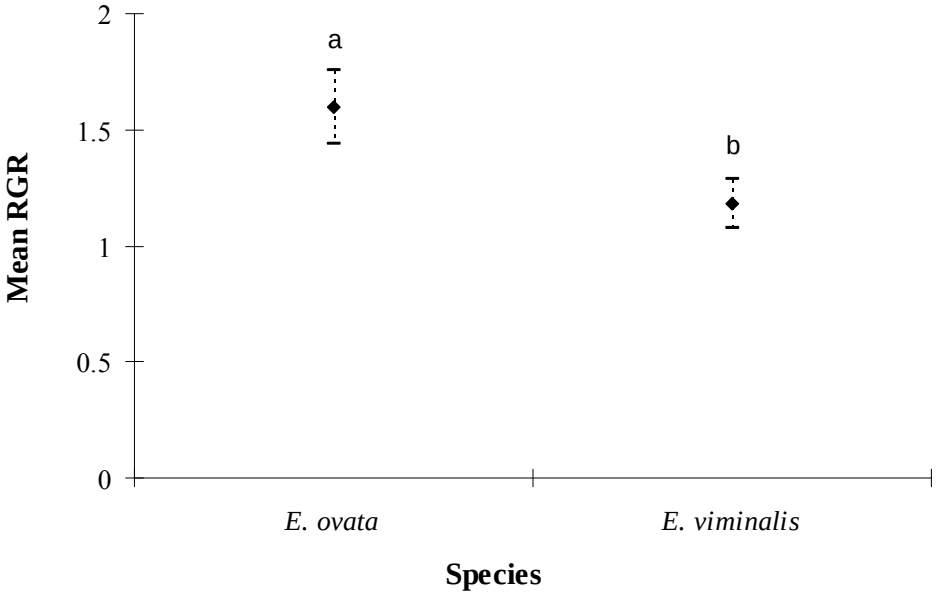


Figure 4.9: The response of *E. ovata* and *E. viminalis* when planted in the *Juncus* weed community. Data points show the mean RGR ± standard error. Means with no letters in common are significantly different at $p < 0.05$ (Tukey HSD test).

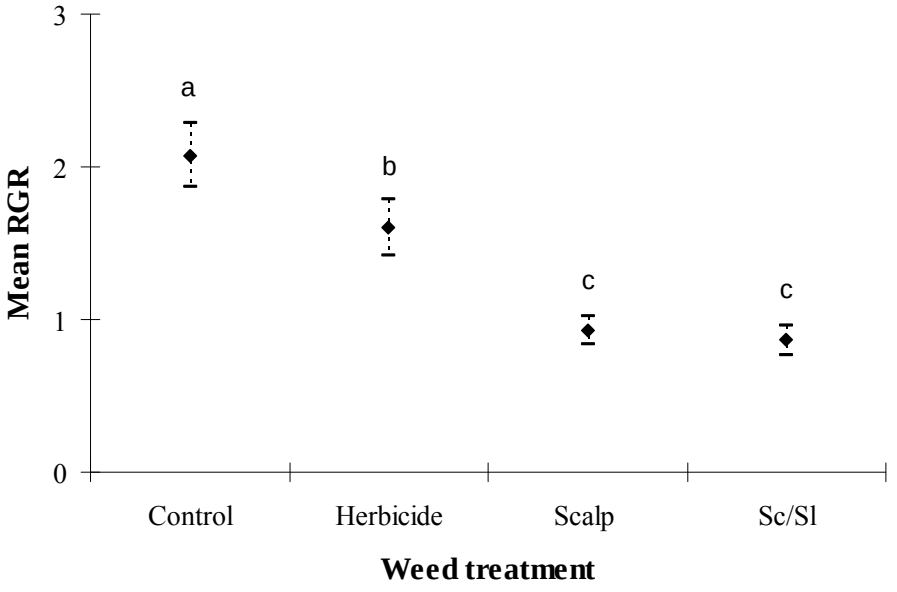


Figure 4.10: The effect of weed treatment on the RGR of *E. ovata* and *E. viminalis* seedlings in the *Juncus* weed community. Data points show the mean RGR ± standard error. Means with no letters in common are significantly different at $p < 0.05$ (Tukey HSD test).

Senecio community

Unlike the results observed in the two other locations, within the *Senecio* community a significant combined effect of treatment and species type occurred (GLM, F = 2.29, p = 0.011), indicating that individual species had significantly different responses to the treatments thus different species preferred different ground preparation treatments for optimum seedling growth (to be discussed in the analyses of individual species response to location and treatment type).

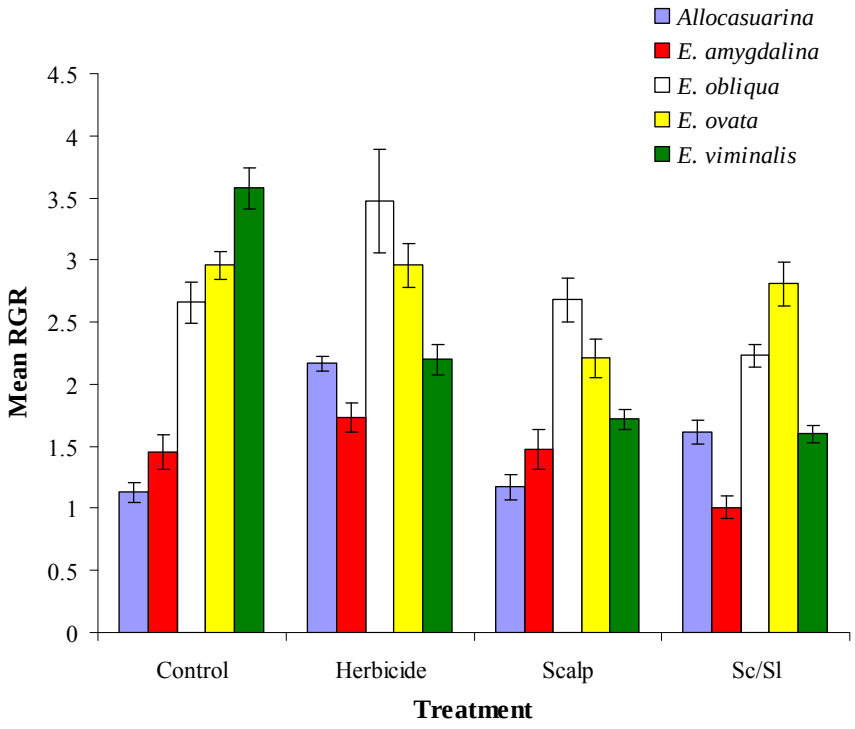


Figure 4.11: The mean Relative Growth Rate (RGR) of five tree species grown under different weed control treatments in the *Senecio* weed community. Columns represent the mean ± standard error.

As well as a significant combined effect, the main effects of species type and treatment were also found to be significant within the *Senecio* location. The three *Eucalyptus* species, *E. obliqua*, *E. ovata* and *E. viminalis,* were found to have significantly greater growth within this habitat than *E. amygdalina* and *Allocasuarina* (Figure 4.12).

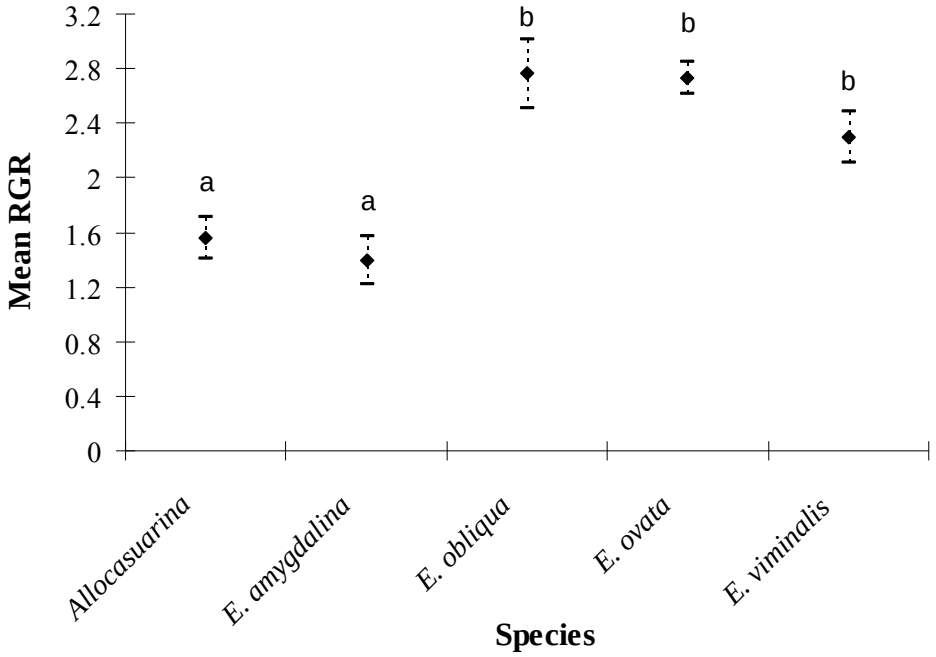


Figure 4.12: The response of five tree species planted in the *Senecio* weed community. Data points show the mean RGR ± standard error. Means with no letters in common are significantly different at $p < 0.05$ (Tukey HSD test).

Pairwise comparisons of the weed treated and control plots (Figure 4.13) showed that seedlings recorded the greatest RGR in the control and herbicide treated plots (2.6 ± 0.20 and 2.5 ± 0.14) which was significantly different to the RGR recorded in the scalp and Sc/Sl plots (1.81 ± 0.16 and 1.90 ± 0.19).

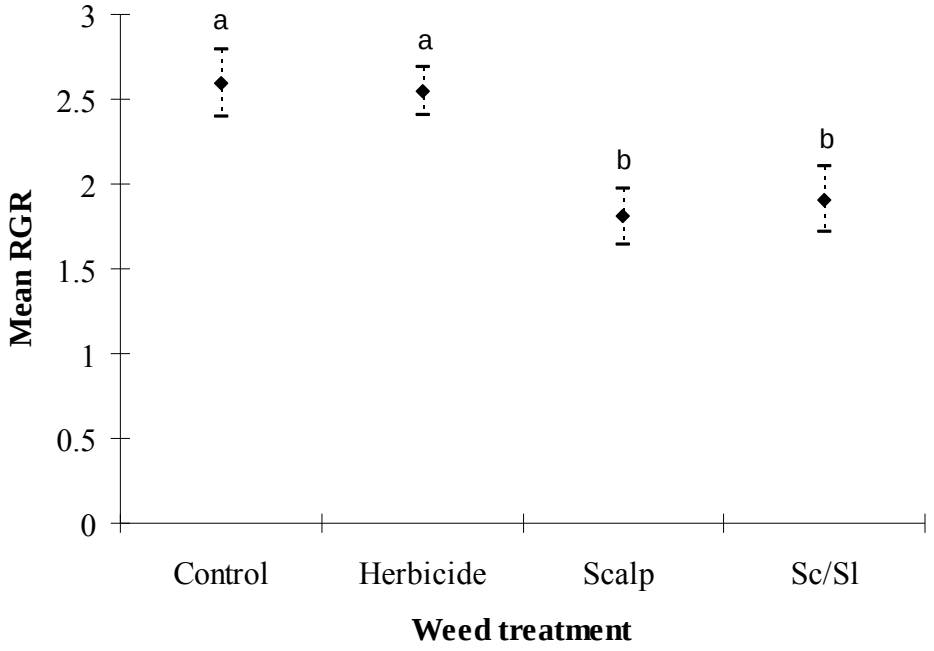


Figure 4.13: The effect of weed treatment on the combined RGR of five tree species planted in the *Senecio* weed community. Data points show the mean ± standard error. Means with no letters in common are significantly different at $p < 0.05$ (Tukey HSD test).

Seedling growth – Effects of weed community and weed treatment on the growth of five tree species.

Allocasuarina littoralis

The growth of *A. littoralis* seedlings was seen to be significantly affected by community (GLM, $F = 45.54$, $p < 0.0005$) as well as the type of weed treatment applied in combination with community (GLM, $F = 3.74$, $p = 0.016$). However, the GLM analysis showed no significant effects of weed treatment alone ($F = 0.78$, $p = 0.513$). From Figure 4.14 it can be seen that *A. littoralis* had the greatest seedling growth in the *Senecio* habitat where the average RGR was significantly higher than the result obtained in the Bracken habitat.

Though the GLM analysis showed that treatment did not have a significant effect on data averaged across the two habitats, the significant result for the combined effect of community and weed treatment indicated that seedling growth had a significant response to weed treatment depending on community. The effect of weed treatment was found to be insignificant in the Bracken habitat but significant in the *Senecio* community as seen in Figure 4.14. In the *Senecio* community the plots that were treated with herbicide had the greatest seedling RGR of 2.17 ± 0.19 which was significantly higher than the RGRs recorded for the control (1.13 ± 0.43) and scalp (1.17 ± 0.21) plots.

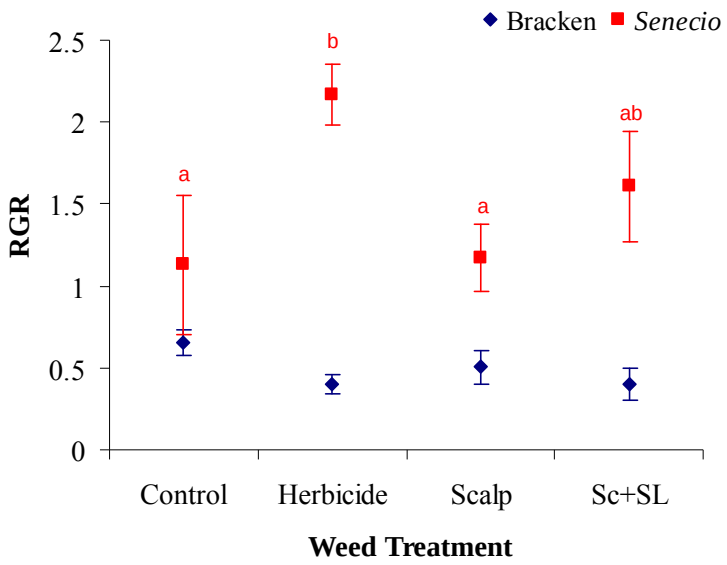


Figure 4.14: The effect of community and weed treatment on the relative growth rate (RGR) of *Allocasuarina littoralis*. Data points show the mean ± standard error. Means with no letters in common are significantly different at $p < 0.05$ (one-way ANOVAs within each community). Note - There were no significant differences between means for weed treatments within the Bracken community.

Eucalyptus amygdalina

Community was the only factor to have a significant effect ($F = 43.05$, $p < 0.0005$) on the growth of *E. amygdalina* seedlings whereas treatment ($F = 1.39$, $p = 0.259$) and the combined effect of treatment and community ($F = 1.42$, $p = 0.251$) were not significant in the GLM analysis (Figure 4.15). The mean RGR of seedlings grown in the Bracken community was 0.35 ± 0.07 which was significantly lower than the mean RGR of 1.392 ± 0.18 calculated for the *Senecio* habitat.

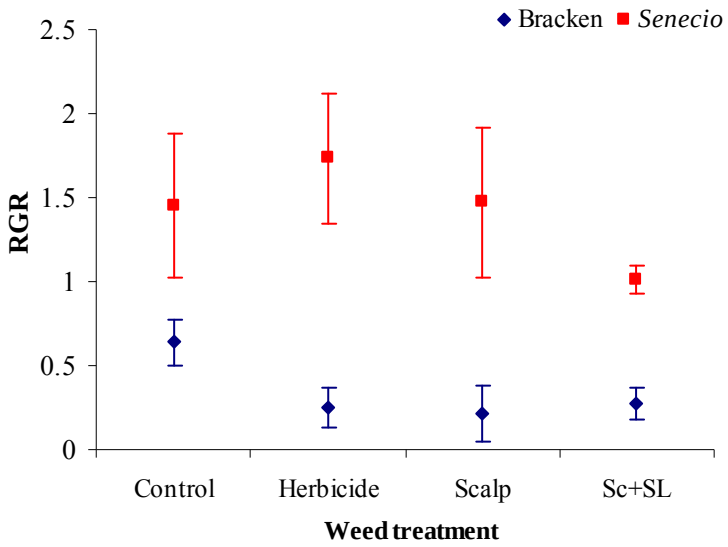


Figure 4.15: The effect of weed community and weed treatment on the relative growth rate (RGR) of *E. amygdalina*. Data points show the mean ± standard error. Note - There were no significant differences between means for weed treatments within each location.

Eucalyptus obliqua

As observed with *A. littoralis* and *E. amygdalina*, seedlings of *E. obliqua* had their greatest growth in the *Senecio* weed community which was significantly different from the growth observed in the Bracken community. There was, however, no significant effect of treatment or combined effect of treatment and community on *E. obliqua* growth (Figure 4.16). The mean RGR observed in the *Senecio* community was 2.80 ±0.25 whereas the mean RGR in the Bracken community was only 0.59 ± 0.11.

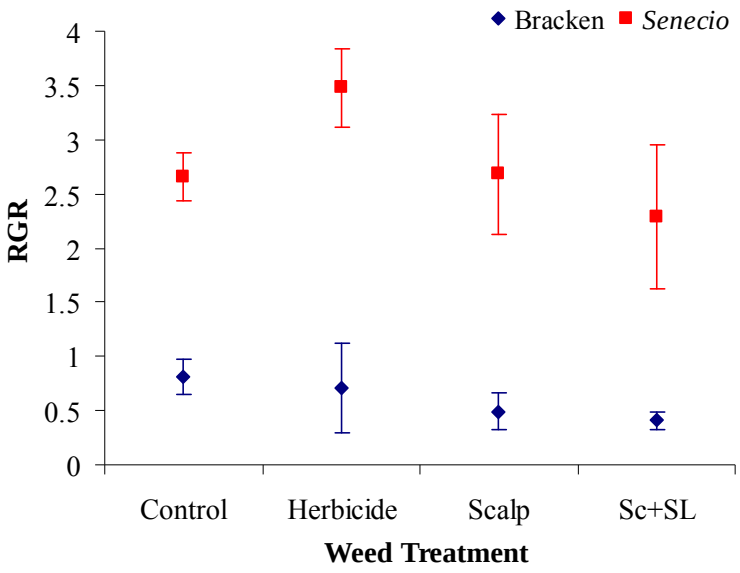


Figure 4.16: The effect of weed community and weed treatment on the relative growth rate (RGR) of *E. obliqua*. Data points show the mean ± standard error. Note - There were no significant differences between means for weed treatments within each location.

Eucalyptus ovata

The growth of *E. ovata* seedlings were seen to be significantly affected by weed community, weed treatment and the combination of these two factors. The mean RGR of 2.73 ± 0.12 obtained in the *Senecio* habitat was greater than that recorded in the two other weed communities. The mean RGR of 1.60 ± 0.16 observed in the *Juncus* habitat was in turn significantly greater than the mean RGR of 0.68 ± 0.08 obtained in the Bracken habitat.

Within the *Senecio* and Bracken communities, weed treatment did not have a significant impact on seedling growth (Figure 4.17). However, within the *Juncus* community there were significant differences between the weed treatments and control plots. The greatest growth of 2.52 ± 0.34 within the *Juncus* community was recorded within the control plots which was similar to the mean RGR of 1.99 ± 0.23 recorded in the herbicide treated plots. The mean RGRs of both the control and herbicide treated plots were significantly greater than the mean RGRs of 1.059 ± 0.11 and 0.84 ± 0.14 recorded in the scalp and Sc/Sl plots respectively.

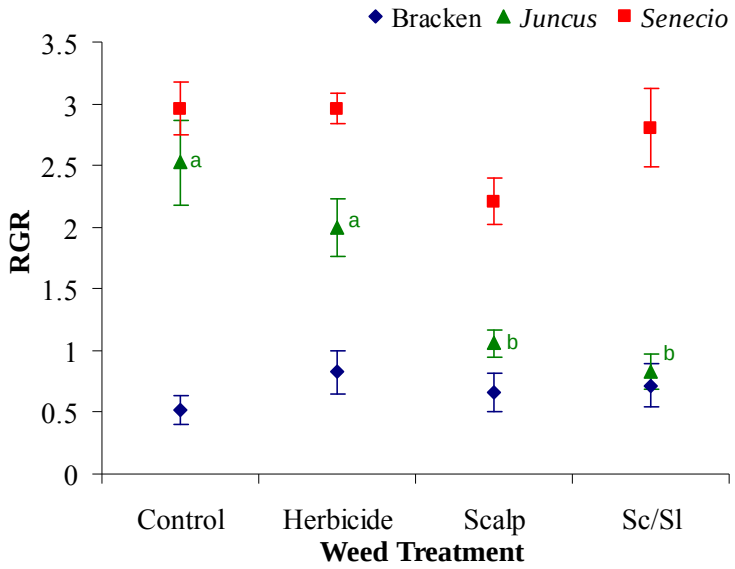


Figure 4.17: The effect of weed community and weed treatment on the relative growth rate (RGR) of *E. ovata*. Data points show the mean ± standard error. Means with no letters in common are significantly different at $p < 0.05$ (one-way ANOVAs within each location). Note - There were no significant differences between means for weed treatments within the *Senecio* and Bracken communities.

Eucalyptus viminalis

E. viminalis seedling growth was significantly affected by community (GLM, $F = 93.25$, $p < 0.0005$) as well as by weed control application (GLM, $F = 15.59$, $p < 0.0005$) but not by the combination of these two factors (GLM, $p = 0.396$, $F = 1.06$). Significant differences in seedling growth were attained in each community with the lowest mean RGR of 0.73 ± 0.07 occurring in the Bracken community while the mean RGR recorded in the *Juncus* community was 1.18 ± 0.10 and the greatest mean RGR of $2.29 \pm .19$ was attained in the *Senecio* community. Within each weed community the effect of weed treatment was similar (Figure 4.18), with seedling growth being significantly greater in the control plots than in the plots that had weed treatments applied.

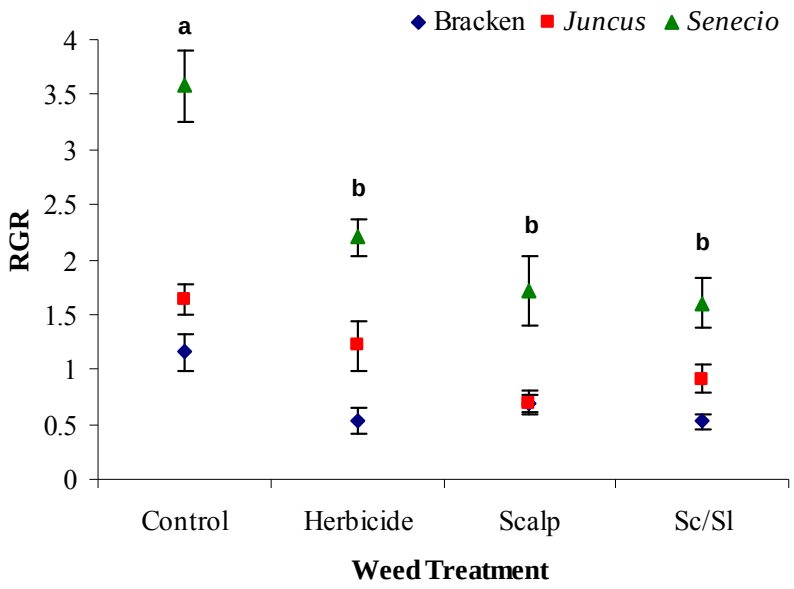


Figure 4.18: The effect of weed community and weed treatment on the relative growth rate (RGR) of *E. viminalis*. Data points show the mean ± standard error. Treatments with no letters in common are significantly different at $p < 0.05$ (Tukey HSD test). Note - as there was an insignificant combined effect of treatment and location, the pairwise comparison of treatment means was undertaken on data pooled across the three locations.

Discussion

Effect of weed control on vegetation composition

After two years, only treatments that incorporated scalping were able to sustain significantly greater bare ground cover than the control plots in the *Juncus* and Bracken communities. The scalping treatments were also found to reduce exotic grass cover in both these communities. These findings reflect other studies (Smith *et al.* 1991; Zacharek 1996; Geeves *et al.* 2008) which have found that weed control treatments which reduce the root mass result in slower recolonisation of exotic grasses. However, within the *Senecio* community neither of the weed control treatments were able to sustain a significant cover of bare ground cover or reduce exotic species cover. This was probably influenced by the fact that the dominant vegetation of this community was the native *Senecio* spp., which did have a significant reduction in cover by all three weed control applications, with the most effective treatment being the combined treatment of scalping and slashing.

Scalping proved to be ineffective in reducing the cover of *Trifolium repens* (clover) within the *Juncus* community, whereas glyphosate application was found to be successful in significantly reducing the cover of clover. This infectiveness of scalping to reduce the cover of clover in old fields was also observed by Zacharek (1996).

The frequency of native species within the *Juncus* community also significantly increased following weed control by scalping, in particular the plot frequency of the native *Isolepis* spp.. *Isolepis inundata* is a commonly observed opportunistic pioneer on acid sulfate soil scalds along the NSW coast where it spreads onto scalds during wet periods retreating when conditions are dry (Rosicky *et al.* 2006). The observations in the present study were recorded during October 2002 when soil moisture levels were significantly higher than during the summer months. Thus, *Isolepis* cover may be found to also retreat in the summer resulting in reduced native cover of the plots.

Due to the rapid recovery of weed species, follow-up weed control is often recommended (Panetta and Groves 1990; Petrov and Marrs 2000; Siipilehto 2001; Campbell and Nicol 2002) for effective reduction in weed cover. The scalping treatment that was followed by slashing 9 months later (Sc/Sl) did prove to be the most effective method for reducing the cover of *Pteridium esculentum* (bracken). Even though *Pteridium esculentum* (bracken) is a native species, a factor considered to effect poor sclerophyll forest establishment on abandoned farmland is competition from bracken (Zeng and Whelan 1993). The effectiveness of Sc/Sl in reducing bracken cover resulted in a significant increase in frequency of native herb species in Sc/Sl treated plots.

Seedling survival and growth

Two years post-planting, *Eucalyptus* seedling survival was not improved by weed control in any of the 3 locations, with glyphosate application only improving the survival of *Allocasuarina* in the *Juncus* location. Although many previous studies have reported that a lack of adequate moisture and grass competition often limit *Eucalyptus* seedling survival (Semple and Koen 1997; Van der Meer *et al*. 1999; Clarke 2002), in the present study landscape position was found to be the most important factor.

Eucalyptus ovata and *E. viminalis* were the only species to successfully establish in the *Juncus* spp. rushland. This community had poor soil drainage and significantly higher soil moisture in all seasons than the *Senecio* and Bracken communities. The reconstruction of the native vegetation of the field site prior to agricultural disturbance (Chapter 2) identified that *E. ovata* woodland occurred in the low-lying *Juncus* spp. community. Thus the successful establishment of *E. ovata* can be attributed to the *Juncus* community corresponding with that of *E. ovata's* typical distribution on lower altitudes (< 200 m) in seasonally waterlogged flats (Ladiges *et al.* 1981). Bell (1999) found that Australian species that are naturally found on river flood plains have inherent capabilities to rehabilitate damaged agricultural catchments.

The ability of *E. viminalis* and *E. ovata* to tolerate waterlogging has been attributed to adventitious roots, stem hypertrophy and the production of aerenchyma tissue that are induced by flooding (Ladiges and Kelso 1977). However Ladiges and Kelso (1977) also found that differences in waterlogging tolerance may exist between the populations of *E. viminalis* native to low lying areas and those that occur on free draining sites. As the drainage conditions at a planting location are critical to seedling health (Crowell and Lane 2001), the careful selection of seedlings whose ecological range and water tolerance corresponds with that of the restoration site has shown to be a significant factor in predicting seedling performance (Van der Moezel *et al*. 1991). Akilan *et al*.'s (1997) field and glasshouse experiments on river red gum (*Eucalyptus camaldulensis*) genotypes also found that the restoration of the water balance of damaged agricultural catchments can be best managed by matching specialized genotypes of river red gums with particular catchment positions.

Seedling growth was likewise strongly influenced by landscape position, with weed control having no effect on growth within the *Juncus* and Bracken communities. Although the percentage of surviving seedlings was high for all species in the Bracken community, seedling growth was the poorest in this location. Unlike the *Juncus* and *Senecio* communities, the Bracken fernland was not improved pasture as indicated by the significantly lower available phosphorus in its soil. Seedlings planted on ex-native sites often show poorer growth than when planted in ex-crop or improved pastures and repetitive weed control until canopy closure has been found to aid seedling performance (Andrews 2000; Adams *et al.* 2003; Close and Davidson 2003; Li *et al.* 2003). However within the Bracken fernland, the greatest seedling growth for all species was achieved when seedlings were planted without weed control. This is in contrast to the findings of Tolhurst and Turvey (1992) who concluded that bracken competition reduced seedling vigour with the number of *Eucalyptus* seedlings established inversely related to bracken leaf area index. However they also found that bracken competition did not differentially affect *Eucalyptus obliqua*, as is the case in the present study. Vesk and Dorrough (2006) also found that native pastures may not need to be treated to promote *Eucalyptus* regeneration.

The improved seedling growth within the Bracken control plots could possibly be due to shading from vegetation reducing soil temperatures and evaporation rates (Hastwell and Facelli 2000; Lloret *et al*. 2005), indicating that the low soil fertility and soil moisture levels contributed to poor seedling growth rather than competition from weeds. The combination of competition from Bracken and poor soil fertility has previously been thought to explain the poor natural regeneration of tree species on abandoned farmland (Zeng and Whelan 1993). However, in the current study the size of seedlings prior to planting may have influenced their ability to compete for nutrients.

In studies on *Eucalyptus globulus* Close *et al*. (2003) found that the size of the root ball at time of planting largely accounted for observed height growth after 6 months and Nuglube (1988) observed that *E. camaldulensis* seedlings that were at least 16-25 cm in height when planted were the most effective for dry zone afforestation in Malawi after 12 months. Furthermore, the usual poor growth of seedlings when planted on abandoned farmland may have been overcome by mycorrhizal development on the seedlings through the addition of forest soil to the potting mix (Ellis and Pennington 1992; Brockwell *et al*. 2011).

Even though the *Juncus* community greatly differed in soil resources to that of the Bracken location, seedling growth of *E. ovata* and *E. viminalis* within the *Juncus* community was also greater when no weed control treatment was applied. When revegetating wetland sites, Raulings *et al*. (2007) found that age of plants and the depth of water were important factors in the survival of swamp paperbark seedlings whereas planting method seemed to have little effect. Thus, the greater growth that was observed in the control plots may be explained by the presence of vegetation to reduce the amount of standing water that seasonally occurred through the cooler months.

An effect of weed control on seedling growth was only detected in the *Senecio* community, with all four *Eucalyptus* species and *Allocasuarina* having the greatest seedling growth within this location. The improved soil nutrient levels at this location due to past land and fertiliser use may override the natural resource patterns of the

landscape (Cole and Lunt 2005) thus promoting the growth of all planted seedlings irrespective of their relative natural distributions. However, due to the location of the *Senecio* community mid-way up the western slopes of the property, the natural distributional range of all the species planted is likely to have corresponded with this location. Only *Allocasuarina* seedling growth benefited from weed control applications with its growth significantly improved in the herbicide treated plots. However, the seedlings planted in the scalp treatments had similar growth to the control. Although, competition from weeds in this community was observed to greatly limit native establishment from seed (Chapter 3), it does not seem to be a limiting factor on seedling survival and growth. As with the Bracken community, the ability of seedlings to grow successfully without weed treatments may be attributed to initial seedling size and mycorrhizal development on seedlings.

Conclusions

Despite the fact that scalping sustained bare ground cover and increased native species richness, weed control treatments did not improve the survival and growth of seedlings. Of greater importance to seedling survival was the association between species natural distribution and the habitat being restored, in particular drainage conditions. Slower growth rates in the Bracken fernland can be attributed to poorer soil conditions though the locally sourced seedlings still grew best without weed treatment, thus indicating that local species can still be used to revegetate depleted soils.

Chapter Five

Intra- and interspecific neighbour interactions among *Eucalyptus obliqua, Allocasuarina littoralis* and *Poa labillardierei*

Introduction

As exotic species frequently have superior colonising abilities to native species, the creation of gaps in the grassy sward is often not enough to promote the establishment of native species (Maron and Connor 1996). Thus, in highly modified landscapes the task of restorationists is often to initiate successional development by replanting with native species (Venning 1988 in Windsor 1999; Yates and Hobbs 1997a; Lamb 1998).

The re-introduction of native tree species usually restores the productive capacity of the landscape but does little to recover biological diversity, as native understorey species are seldom used in restoration plantings or sowings (Lamb 1998; Windsor *et al*. 1999). The restoration of understorey species is usually left to secondary succession. Little is known about the time period needed to achieve a community species pool with a richness equal to the remnant forest. For this reason, the planting of species mixtures of trees, shrubs and grasses has been recommended (Windsor *et al.* 1999). It is also important for sites to be managed in ways that minimise opportunities for invasion of exotic species (Maron and Connors 1996). Windsor *et al*. (1999) found that weed control appears to be essential in achieving the objective of recreating *Eucalyptus* woodland and that the abundance of exotic species decreased with higher local native ground cover.

The potential advantage of increased understorey diversity could be offset by a reduction in overstorey tree growth (Lamb 1998). However, mixing trees with species of contrasting phenologies or with different root and canopy architectures, as well as using

nitrogen fixing species, could be ways to promote production in the overstorey (Lamb 1998). The use of facilitative interactions between a nurse-plant and the plant species of interest has been proposed as a successful restoration technique in stressful environments, such as deserts, salt marshes and alpine communities (Rudgers and Maron 2003; Pueyo *et al*. 2009). Positive associations have also been reported between trees and shrubs in woodland and grassland environments, where shrubs have been found to facilitate growth of pine, *Quercus douglasi,* seedlings (Callaway 1992); and between shrubs and herbaceous plants in Florida scrub (Mondo *et al*. 2010). In these cases, woody shrubs facilitated seedling establishment through providing shade.

Nurse plants may also ameliorate abiotic stress by lowering soil surface temperatures, decreasing evaporation, offering protection from herbivores, reducing competition with herbs or increasing soil nutrients beneath their canopies (Greenlee and Callaway 1996; Roussett and Lepart 1999; Gomez-Aparicio *et al*. 2005a; Maher *et al*. 2010). However, under some conditions, competition between nurse plants and seedlings could exceed facilitation. Grime (1977) suggests that competition becomes less important in stressful environments. Callaway (1998) also hypothesized that if severe physical conditions restrict resource acquisition, amelioration of this stress by a neighbour may be more likely to favour growth than competition is likely to reduce growth. In east African savannas, Ludwig *et al*. (2001) observed that *Acacia tortilis* trees had a positive effect on grass production in the dry season when water was scarce, but a negative impact during the wet season.

As well as being influenced by abiotic variation, facilitative interactions are influenced by herbivory, life stage, size, growth form and the physiology of the neighbouring plant (Callaway 1998; Munzbergova and Ward 2002; Riginos and Young 2007). Regeneration of many species is facilitated by adult plants which can alter both the above ground environment (canopy effect) and below ground (soil effect) environment to benefit seedling growth (Gomez-Aparicio *et al.* 2005b).

The majority of investigations on facilitative interactions to date have focused on seedlings of the beneficiary species and adults of the benefactor (Rudgers and Maron 2003). However, Kochy and Wilson (2000) investigated whether the seedlings of different herbaceous growth forms would have the same competitive effect i.e. whether woody seedlings would have little competitive advantage over grasses. Their results suggested that interactions between seedlings of different growth forms are symmetric during the early stages of growth but woody seedlings become more competitive as their roots extend beyond the grass rooting layer.

In the few experiments testing interactions among transplanted seedlings in temperate environments, seedling performance in small soil disturbances in prairie grasslands was shown to be unaffected by the number of neighbours but strongly affected by the identity of neighbours (Rabinowitz and Rapp 1985; McConnaughay and Bazzaz 1990). It appears that some combinations can result in positive interactions and thus have good "ecological combining abilities" (McConnaughay and Bazzaz 1990). For example, in their study of regeneration trends of cool temperate Tasmanian rainforests, Hickey and Wilkinson (1999) found that the height growth of *Acacia melanoxylon* was improved by planting in mixed species stands.

Whether planting seedlings at high density with conspecifics would increase their ability to compete against established prairie grasses has also been investigated by Li and Wilson (1998) for both a shrub and tree species. They found that planting the shrub *Symphoricarpos occidentalis* at high density had positive effects on the growth of conspecifics without significantly reducing grass mass because the different growth forms had the same competitive effect. In contrast, growth of the tree *Picea glauca* was not facilitated at high density in the presence of grass, possibly because intense intraspecific competition may have outweighed any competitive effects of the grass mass.

Few studies, however, have looked at the facilitative effects of planting mixtures of two or more species or have measured whether planting mixtures of species accelerates

recovery sufficiently over the long-term to warrant this level of intervention. This makes it difficult to predict the impact of different planting designs on facilitating seedling growth and reducing non-native cover (Corbin and Holl 2012).

This chapter investigates whether the recovery of *E. obliqua* open-forest on abandoned farmland can be accelerated by the simultaneous re-introduction of tree, shrub and groundcover species. Information on the interactions of *E. obliqua* with neighbouring plants mainly comes from silvicultural studies (Bi *et al.* 1992; Bi and Turvey 1994a; Bi and Turvey 1994b), in which the effects of *E. obliqua* competition on plantation trees such as *Pinus radiata* are assessed. Little is known about interactions between *E. obliqua* seedlings and native shrub and groundcover species. As *Allocasuarina littoralis* and *Poa labillardierei* are both common species found in the *E. obliqua* open forest, they have been selected to be planted in mixtures with *E. obliqua* to assess the performance of planting mixtures of tree, shrub and groundcover species in accelerating native plant restoration.

In the remnant forest of the study site, the overstorey is dominated by *E. obliqua* with *A. littoralis* present but not dominant in the shrub layer (see Chapter 2 for floristic descriptions of vegetation communities) which indicates a mutual co-existence and potential for simultaneous reintroduction. The nitrogen-fixing ability of *A. littoralis* indicates that it may have a beneficial effect on neighbour plants (Mowry 1933; Binkley 1992). Facilitation by the nitrogen fixing species *Acacia mearnsii* resulted in 50:50 mixtures of *E. globulus* and *A. mearnsii* having above ground biomass production twice that of *E. globulus* monocultures (Forrester *et al.* 2005). However with high levels of nitrogen fertiliser, *E. globulus* suppressed *A. mearnsii* growth and biomass production of mixtures was not significantly different to that of *E. globulus* monocultures. Forrester *et al*. (2005) suggested that "mixtures should only be planted on sites where interactions between species will increase the availability of, or reduce competition, for a major limiting resource for growth at the site".

Information on the effect of grasses on woody plants is also not well understood as these interactions are largely below ground (Peltzer and Kochy 2001). Although *P. labillardierei* is the dominant grass in the *E. obliqua* open forest there is evidence that it may inhibit eucalypt seedling growth (Fensham and Kirkpatrick 1992). The ungrazed *P. labillardierei* sward was found to inhibit *Eucalyptus rodwayi* germination and seedling growth which was likely to have been the result of physical resistance of the grass root mat to root penetration by *E. rodwayi.*

As well as the identity of neighbour plants, growth is closely related to the distance of neighbours. The presence of neighbours can in turn have contrasting effects on height and biomass (McConnaughy and Bazzaz 1990). The aims of the present study are to test:

1. If seedling survival and growth is affected by identity and combination of neighbour type?
2. If there is a combined effect of neighbour combination and distance of nearest neighbour?.
3. If plant height and biomass are affected differently by neighbour type?

Thus, this chapter aims to determine whether the simultaneous reintroduction of tree, shrub and ground cover species could accelerate dry sclerophyll forest restoration on agricultural land compared to plantings of single species by examining the competitive effects of *Eucalyptus obliqua* seedlings, *Allocasuarina littoralis* seedlings and *Poa labillardierei* tillers.

Materials and methods

Plant material

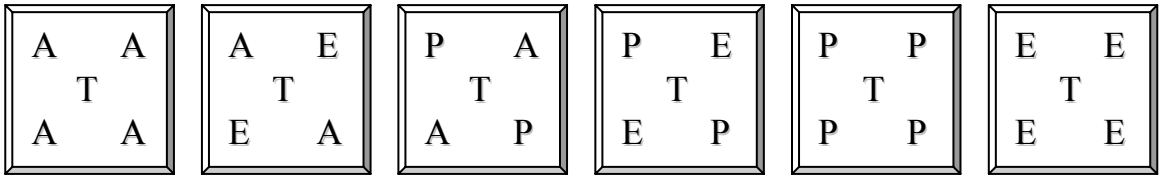
The seed of *Eucalyptus obliqua* and *Allocasuarina littoralis* was collected in May and June 2000 from within 10 km of the study site at Saltwater River in south-eastern Tasmania. In January 2001, seeds were sown in a seed raising mix of sand, peat and vermiculite (1:1:1) in seedling trays placed within a shade house at the University of Tasmania Hobart campus. Once seedlings had cotyledons they were pricked out into tubes which were 50 mm × 50 mm square and 120 mm deep. *Poa labillardierei* tussocks were collected from the field site between 15th August and 6th October 2000 and sown into pots between the 9th and 19th October 2000. The tussocks were then separated a month before planting to produce the necessary numbers of tillers for the experiment. The potting mix was composed of composted pine bark, sand, peat and A1 horizon loamy sand soil collected in the remnant forest at the study site in the respective ratio of 6:4:1:2, with the potting mix having a pH range between 6.5 –7. The site soil was used in an attempt to introduce mycorrhizal fungi that may assist in the growth of the seedlings. Slow release Osmocote was added to the potting mix in the concentration of 300 g for every 50 L of potting mix.

Planting design

A modified target-neighbour experimental design was used to examine the competitive effects of *E. obliqua* seedlings, *A. littoralis* seedlings and *P. labillardierei* tillers on each other. Monocultures or two species mixtures of four individuals were planted around each target species at two distances: 3.5 cm, labelled the close treatment; and 7 cm, labelled the wide treatment (Figure 5.1). These distances were chosen so that the interaction between the seedlings could be measured in the short-time frame of the study as plants at high density experience stress or benefits from neighbours early in their development (Harper 1977; Li and Wilson 1998). The first distance of 3.5 cm was used as this was the closest distance that the tube stock could be planted next to each other due to the root mass. The second distance of 7 cm was then chosen to measure the extent

of plant interactions at half the first density. In addition each species was also planted without any neighbours, labelled the single planting treatment, so in all, target plants were sown over three densities.

a.



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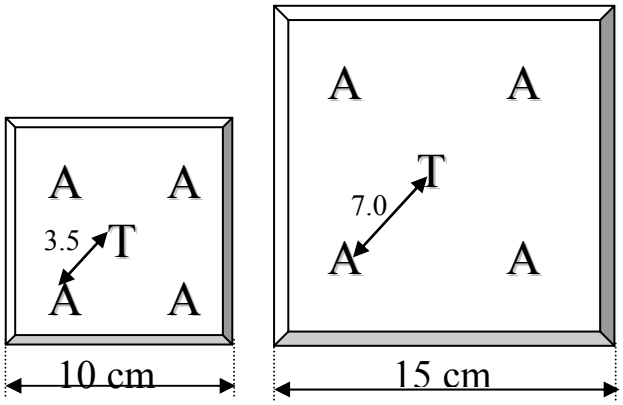


Figure 5.1: Target-neighbour experimental design. a. Each target species was sown with the 6 neighbour combinations at b. two distances. A= *Allocasuarina littoralis*, E = *Eucalyptus obliqua*, P = *Poa labillardierei*, T = Target species (there were six replications of A, E or P as the target species).

Ground preparation and plot layout

The experiment was located in *Senecio* open herbfield (see Chapter 2) in the area of this vegetation type that had the gentlest slope. In September 2000 the vegetation was scalped using a bobcat, removing all above ground weeds in an area of 15 m × 15 m. To protect the seedlings from grazing, a fence which excluded wallabies, rabbits and possums was constructed around the perimeter of the scalped area. Two days prior to planting, experimental plots within this area were once again cleared of all above ground weed matter using a mattock. The plot dimensions were 0.35 m × 3.6 m for the single and close treatments and 0.35 m × 4.6 m for the wide space treatment, allowing for at

least 10 cm between treatments and plot edges for the close and wide treatments and 20 cm for the single plantings. There were 6 replications of each treatment and the plots were positioned using a stratified random experimental design, with each treatment being present in each one of the six rows.

Thus, there was one plot pre row containing;

- six individuals of each species planted 20 cm apart;
- the six treatment types with either *Allocasuarina*, *Eucalyptus* or *Poa* (Figure 5.1a) as the target plant (i.e. 18 treatments in all) with neighbours planted at a distance of 3.5 cm,
- the same six treatments but with neighbours planted at a distance of 7 cm.

The position of the plots in the rows and the treatments within the plots was randomly assigned.

Seedlings and *Poa* tillers were planted in December 2001 and each treatment was watered with 0.5 L water. Any seedlings that died within the first two weeks of the field experiment were replaced as these individuals were deemed to have died as a result of environmental rather than experimental conditions. However, after two weeks all individuals that died were left in place. Once the seedlings and tillers were planted under the experimental conditions described above weeds were continuously removed from experimental plots for the six months that plant growth was recorded. During the summer months watering took place as was deemed necessary to keep the seedlings alive.

Soil properties

Ten soil core samples were randomly collected from non-treatment areas in the 15 m × 15 m experimental site for soil nutrient analysis. The soil cores were 10 cm deep and 5 cm in diameter.

The samples were air-dried and then coarsely sieved to remove roots, litter and stones. The following chemical analyses were undertaken:

- available phosphorus using the Bray extractable method (Jackson 1958);
- total phosphorus using the Perichloric/Nitric/Sulphuric acid digestion method (Allen 1974);
- total nitrogen using the Kjeldahl method (Jackson 1958);
- total organic carbon using the Walkley and Black method (Rayment and Higginson 1992).

Seedling growth analysis

To assess individual plant performance, height (H), maximum width (W_1) and W_2, the width perpendicular to the maximum width (Figure 5.2) was measured for each target and neighbour plant at the start of the experiment and 6 months later. The number of leaves on *Eucalyptus* seedlings was also recorded.

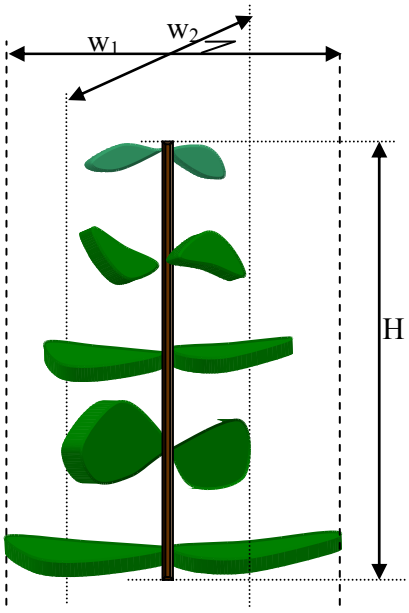


Figure 5.2: Measurements taken to assess plant performance. H = maximum stem height, W_1 = maximum width of plant and W_2 = width perpendicular to the maximum width.

Height analysis

To assess the effect of the competition treatments on target plant height, the maximum stem height was used to calculate the relative growth rate ($RGR_{(H)}$, Equation 5.1a). To assess the $RGR_{(H)}$ of target plants grown with competition in relation to plants grown without neighbours equation 5.1b was applied. The calculated values from equation 5.1b were used to analyse the effects of the six treatment types and distance, as well as a combination of these two factors, on target plant performance (General Linear Model [GLM] analysis, $p < 0.005$).

Equation 5.1

a. RGR_H = (Height at sixth month – Initial height)/Initial height
b. $RGR_{(competition)} - RGR_{(no\ competition)}$

Heights of neighbour plants at initial planting time and at 6 months after treatment were used as co-variates in the GLM. Within the GLM, pairwise comparisons were undertaken using the Tukey Honest Significance Difference (HSD) test at the 95% confidence level. The normality of the data was tested using plots of the standardised residuals versus fitted values and normal probability plots. To meet the assumptions of normality it was necessary to transform some of the data. Square root (RGR + 0.5) or log (RGR + 0.5) were used.

Yield Index analysis

As well as analysing the RGR of target plants using stem height, the effect of the competition treatments on target plant biomass was analysed using a Yield Index that was calculated using the three variables that measured the geometric volume of the seedlings. W_1 (maximum width of plant) and W_2 (width perpendicular to the maximum width) were used to calculate the base area of the seedlings using the ellipse formula:

Equation 5.2

$$A = W_1/2 \times W_2/2 \times \pi$$

The Yield Index (YI) was then calculated using the volume formula of a cone with an ellipse shaped base:

Equation 5.3

$YI = \frac{1}{3}H\ A$

To validate the use of the YI, the relationship between actual yield and the YI was assessed using data from one quarter of all seedlings used in the experiment which were harvested after the 6 month growing period. After harvesting, the seedlings were separated into above ground and below ground material and all soil was cleaned from the shoot and roots. The shoots of harvested seedlings were oven-dried at 105 ^{o}C for three days and then weighed. The regression analysis between actual shoot yield and YI was determined for each species. To meet the assumptions of the regression analysis it was necessary to log transform the data (log (Shoot weight) = $c + b$ log Y.I.). Examination of the normality of the residuals from the regression analyses was also undertaken using the Anderson-Darling normality test at $\alpha = 0.05$.

To determine if a there was a significant effect of competition treatments on target plant yield, GLM analyses were carried out using the same methods as the height analysis, with the relative growth rate calculated using the YI instead of height (Equation 5.4). These analyses were also carried out on the YI of the six neighbour treatments to investigate the response of neighbours to target plant identity and distance.

Equation 5.4

a. RGR_{YI} = (YI at sixth month – Initial YI)/Initial YI
b. $RGR_{(competition)} - RGR_{(no\ competition)}$

Results

Site information

The nutrient status of the soil samples taken from the experimental plots is summarised in Table 5.1. Also included is the soil chemistry data of the soil that was collected from the remnant *E. obliqua* open forest at the study site and the reference values for *E. obliqua* sclerophyll forest as recorded in the Forest Soils of Tasmania (Grant *et al.* 1995). The greatest difference observed between the experimental site and that of soil in the native vegetation communities was the level of total phosphorus. The values for percentage of nitrogen and carbon as well as the pH levels of the experimental site are a close approximation to those found in native forest soils.

Table 5.1: Mean soil nutrient levels ± standard error (n = 10) within the experimental site (area 15 m × 15 m) and reference values for remnant forest soil properties. * indicates missing data.

Nutrient	Total P (ppm)	Available P (ppm)	Total N (%)	Total C (%)	pH	Bulk density
Experimental site (n = 10)	195.7 ± 60.9	22.60 ± 1.71	0.15 ± 0.01	4.26 ± 0.11	4.62 ± 0.01	1.03 ± 0.03
E. obliqua open forest of site (n = 1 composite from 5 samples)	*	5	0.13	*	4.83	*
E. obliqua sclerophyll forest on Triassic sandstone (Ref: Forest soils of Tasmania)	59	*	0.13	4.6	3.9	0.8

During the experimental time-frame greater than average rainfall occurred during the first and last two months (Figure 5.3). The months from February 2000 to April 2000 received less than average rainfall.

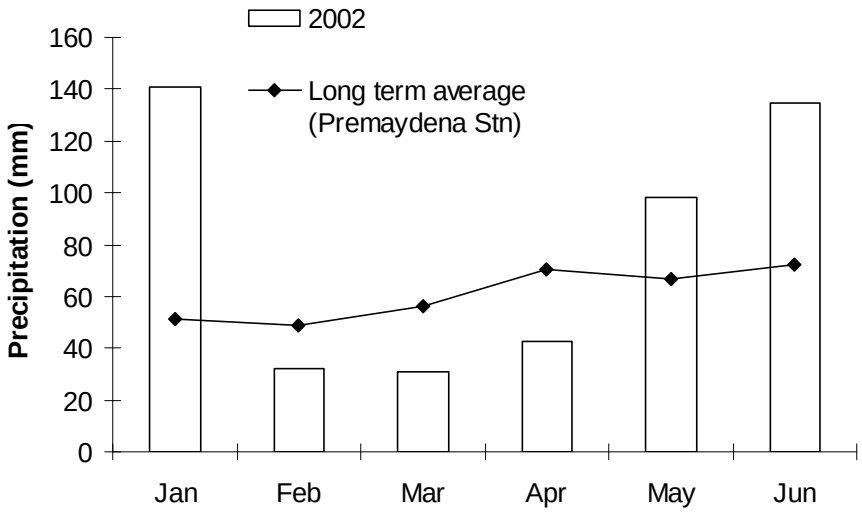


Figure 5.3: The monthly rainfall for Saltwater River in 2002 and the long term monthly average (Bureau of Meteorology).

Target plant stem height response

i. *Allocasuarina littoralis*

After six months the average relative stem height growth ($RGR_{(H)}$) of *Allocasuarina littoralis* seedlings when grown without neighbours was 1.33 ± 0.12, with this range shown by the 2 broken lines on Figure 5.4. As indicated by Figure 5.4, the 4A competition treatment at both the 3.5 cm and 7 cm distance produced significantly greater target plant growth than that of plants grown without neighbours.

For *Allocasuarina littoralis* seedling growth, the effect of treatment was significant (GLM, $F = 10.23$, $p < 0.0005$) but not the effect of distance (GLM, $F = 0.01$, $p = 0.937$) or the interaction of the two factors (GLM, $F = 1.13$, $p = 0.355$). The Tukey HSD test undertaken to assess the effect of treatment on data pooled from the two distances also produced a significant result for the 4A treatment which had a significantly greater mean target plant $RGR_{(H)}$ than all treatments except the 2A + 2E treatment, which produced the next best performance by a target plant. However, the 2A + 2E treatment did not significantly differ from the four remaining treatments (2A + 2P, 2E + 2P, 4E and 4P), which all had a significantly similar target plant $RGR_{(H)}$.

The relationship between target plant performance ($RHR_{(H)}$) and the height of its neighbours at both the initial planting time (GLM, $F = 0.06$, $p = 0.805$) and 6 months after planting (GLM, $F = 0.01$, $p = 0.946$) was also found to be insignificant.

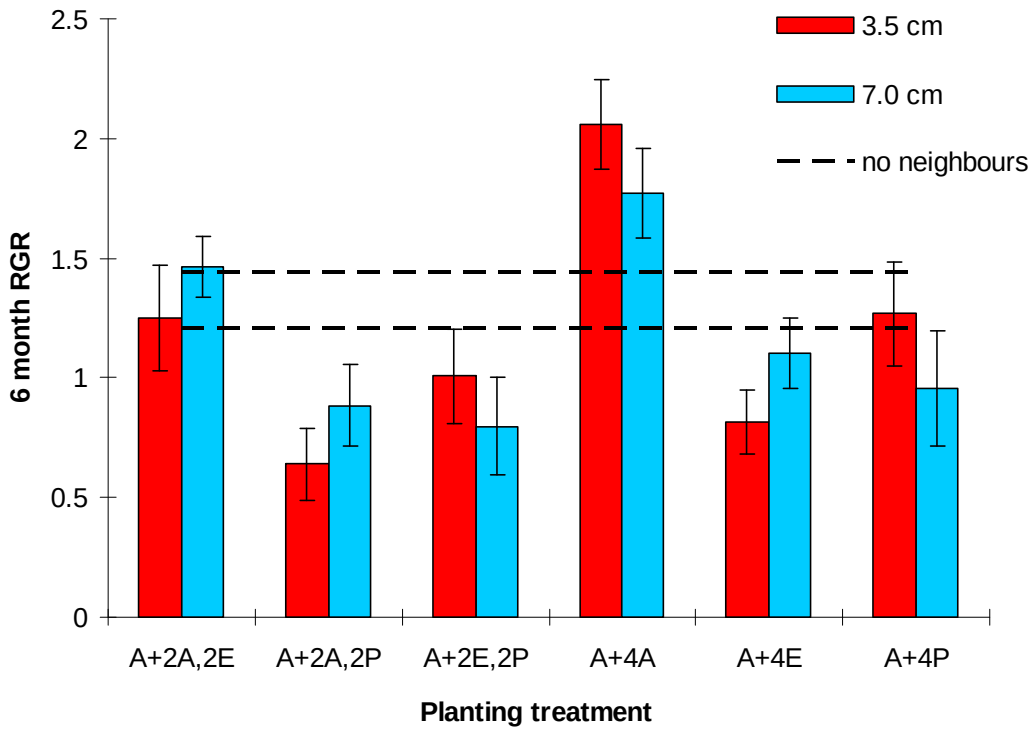


Figure 5.4: $RGR_{(H)}$ of *Allocasuarina littoralis* planted with six combinations of neighbours at 2 distances. Columns show the mean $RGR_{(H)}$ over 6 months ± standard error. The two broken lines indicate the range of the mean $RGR_{(H)}$ ± one standard error when seedlings were grown without neighbours. A = *A. littoralis*, E = *E. obliqua*, P = *P. labillardierei*.

ii. *Eucalyptus obliqua*

The mean $RGR_{(H)}$ of *E. obliqua* seedlings when grown without neighbours after 6 months was 0.62 ± 0.05 which was significantly less than the mean $RGR_{(H)}$ of target plants grown in the following treatments at the close distance; 2A + 2E, 4A and 4E. The 4A treatment at the wide distance also produced a significantly greater target plant mean $RGR_{(H)}$. All other treatments did not have mean $RGRs_{(H)}$ that significantly varied from the seedlings grown without neighbours (Figure 5.5).

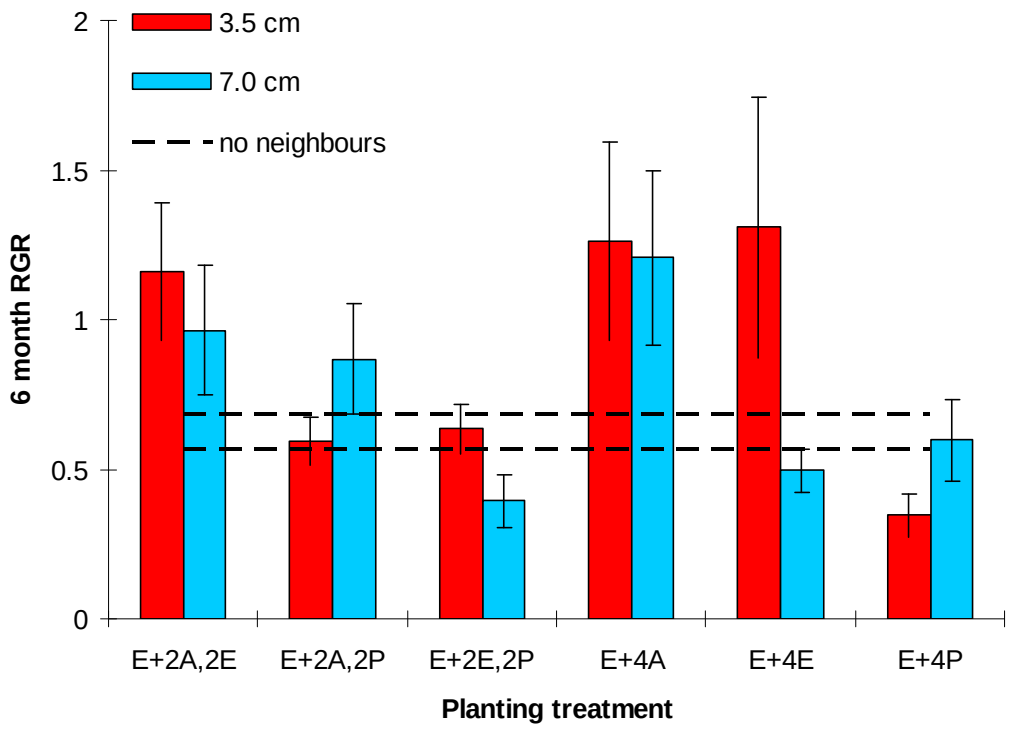


Figure 5.5: $RGR_{(H)}$ of *Eucalyptus obliqua* planted with six combinations of neighbours at 2 distances. Columns show the mean $RGR_{(H)}$ over 6 months ± standard error. The two broken lines indicate the range of the mean $RGR_{(H)}$ ± one standard error when seedlings were grown without neighbours. A = *A. littoralis*, E = *E. obliqua*, P = *P. labillardierei*.

A significant combined effect of treatment and distance (GLM, F = 2.68, p = 0.032) was detected on the mean $RGR_{(H)}$ of target plants, indicating that *E. obliqua* seedlings respond not only to identity of the neighbour plant but also to the distance of neighbouring plants. For example, when the target *E. obliqua* plant was planted in a monoculture with four *E. obliqua* neighbours at a distance of 3.5 cm it had a greater mean $RGR_{(H)}$ (1.31 ± 0.44) than when planted without neighbours (0.62 ± 0.05). However, when the monoculture was planted at a distance of 7 cm, the mean $RGR_{(H)}$ of the target plant (0.50 ± 0.07) was less than that of the plants without neighbours as shown in Figure 5.3. Distance of neighbour plants alone did not have a significant effect on target plant growth (GLM, F = 1.61, p = 0.210) as the mean $RGR_{(H)}$ of 0.87 ± 0.11 of the close treatments did not significantly vary from the mean $RGR_{(H)}$ of 0.74 ± 0.8 of the wide treatments. As found with *Allocasuarina littoralis*, the initial height of

neighbours (GLM, F = 0.10, p = 0.757) as well as neighbour height at 6 months (GLM, F = 2.10, p = 0.153) also had an insignificant effect on *E. obliqua* growth.

iii. *Poa labillardierei*

The mean $RGR_{(H)}$ of *P. labillardierei* tillers when planted without neighbours was 1.21 ± 0.08 which was only greater than the target plant $RGR_{(H)}$ of the 2E + 2P treatment at a distance of 7.0 cm (Figure 5.6). The target plants within three treatments, 2A + 2P, 2E+2P and 4A at the distance of 3.0 cm had considerably greater mean $RGR_{(H)}$s than that of plants grown without neighbours. When neighbour plants were planted at a distance of 7.0 cm from the target plant the treatments 2A + 2P, 4A and 4E produced a greater mean $RGR_{(H)}$.

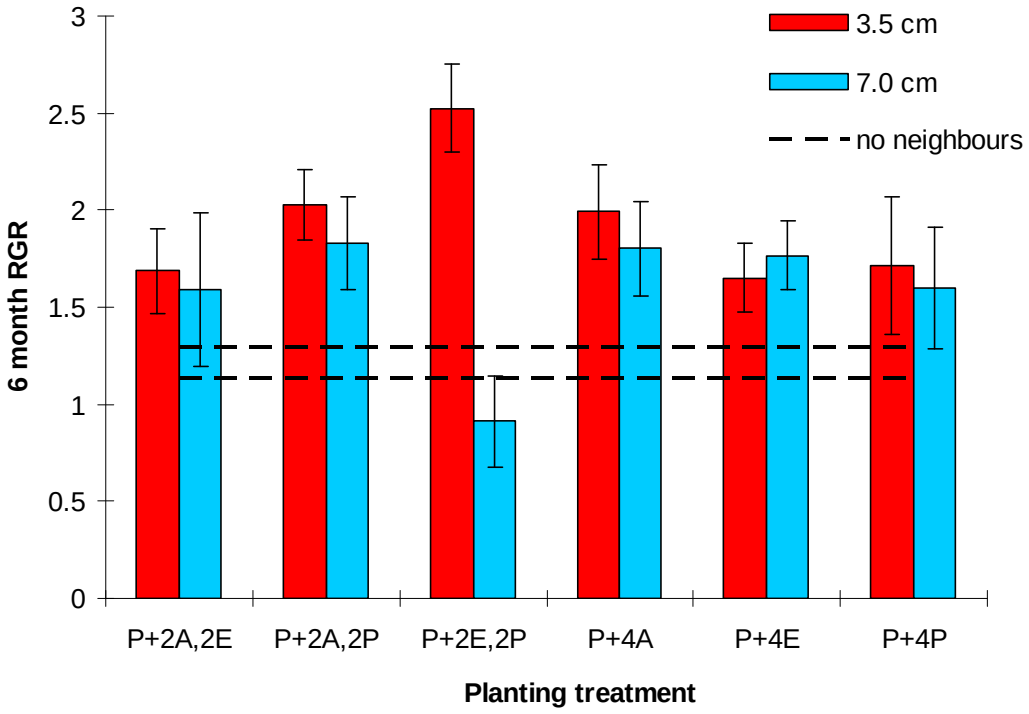


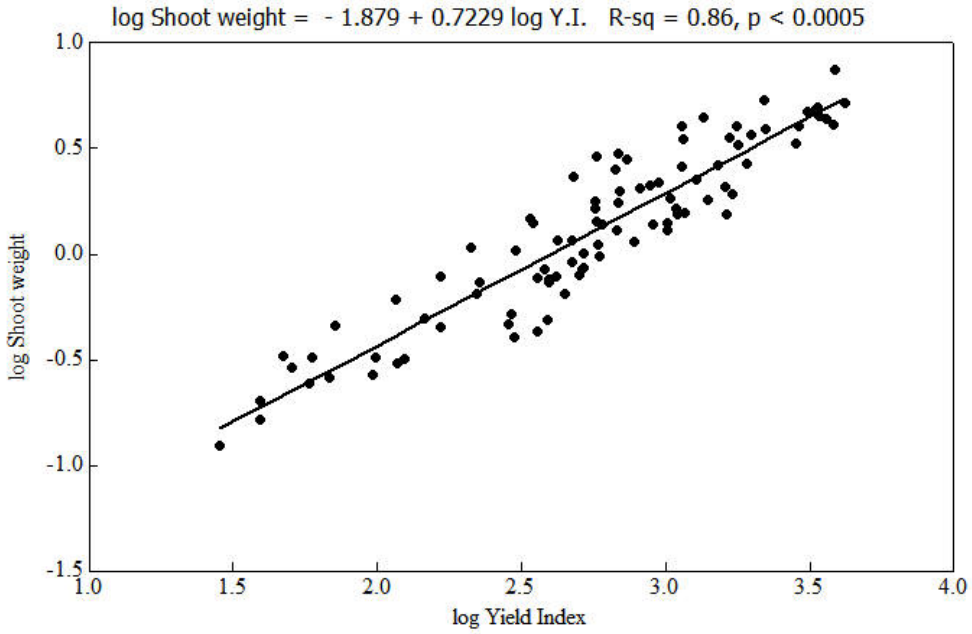
Figure 5.6: $RGR_{(H)}$ of *Poa labillardierei* planted with six combinations of neighbours at 2 distances. Columns show the mean $RGR_{(H)}$ over 6 months ± standard error. The two broken lines indicate the range of the mean $RGR_{(H)}$ ± one standard error when seedlings were grown without neighbours. A = *A. littoralis*, E = *E. obliqua*, P = *P. labillardierei*.

For *P. labillardierei* tillers the effect of distance (GLM, $F = 4.73$, $p = 0.034$) and the combined effect of distance and treatment (GLM, $F = 2.54$, $p = 0.038$) had a significant impact on target plant growth. The mean $RGR_{(H)}$ of all target plants grown when neighbours were 3.5 cm away was 1.92 ± 0.10 whereas the mean $RGR_{(H)}$ of target plants that had neighbours at a distance of 7 cm was 1.60 ± 0.12. The treatment 2E + 2P had a different effect on target plant growth depending on the distance of neighbouring plants. When the *E. obliqua* and *P. labillardierei* plants were at a distance of 7 cm from the target plant the mean $RGR_{(H)}$ of the target plant was 2.03 ± 0.18 which was more than double the mean $RGR_{(H)}$ of 0.91 ± 0.24 which resulted when the neighbours plants were at a distance of 3.5 cm. Although the distance of neighbouring plants had a significant impact on *P. labillardierei*, the height of neighbours did not at both the initial time of planting (GLM, $F = 2.14$, $p = 0.144$) and 6 months after planting (GLM, $F = 1.58$, $p = 0.214$).

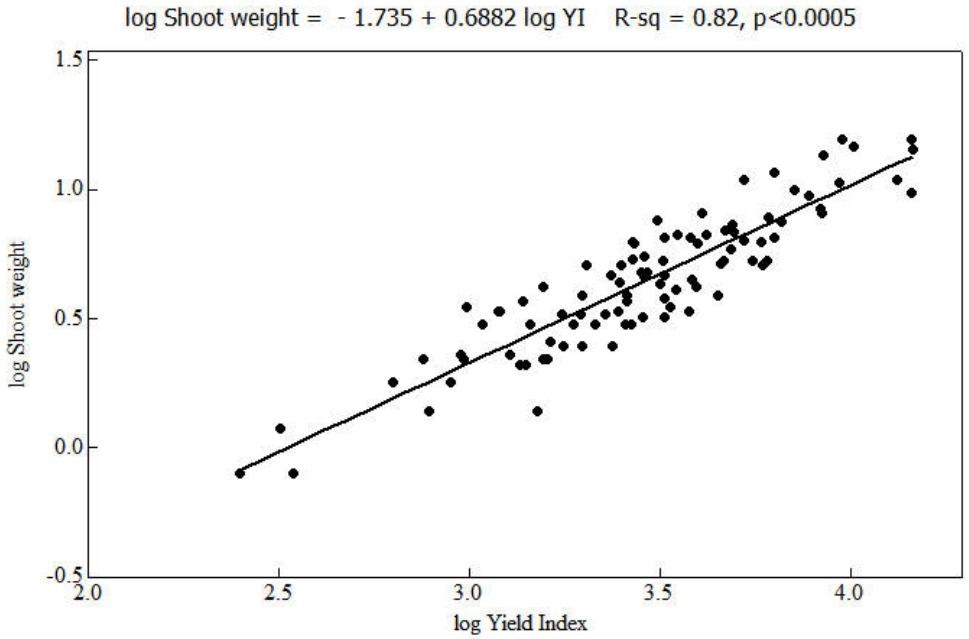
Relationship between Yield Index and shoot weight

For all three species a significant relationship was found between YI and shoot weight using log transformed data (Figure 5.7). The strongest relationship between YI and shoot weight was found for *Allocasuarina littoralis* seedlings ($R^2 = 0.86$, $p < 0.0005$, Figure 5.7a). The YI of *Eucalyptus obliqua* seedlings was also strongly correlated ($R^2 = 0.82$, $p < 0.0005$, Figure 5.7b) to shoot weight. For both the *Allocasuarina* and *Eucalyptus* seedlings a linear regression equation described the relationship between log shoot weight and log YI whereas for *Poa labillardierei* tillers a quadratic equation produced the strongest relationship between log shoot weight and log YI ($R^2 = 0.68$, $p < 0.0005$, Figure 5.7c). However the linear relationship between log shoot weight and log YI of the *P. labillardierei* tillers was not a much weaker correlation, $R^2 = 0.66$, $p < 0.0005$, than that determined using the quadratic relationship.

a.



b.



c.

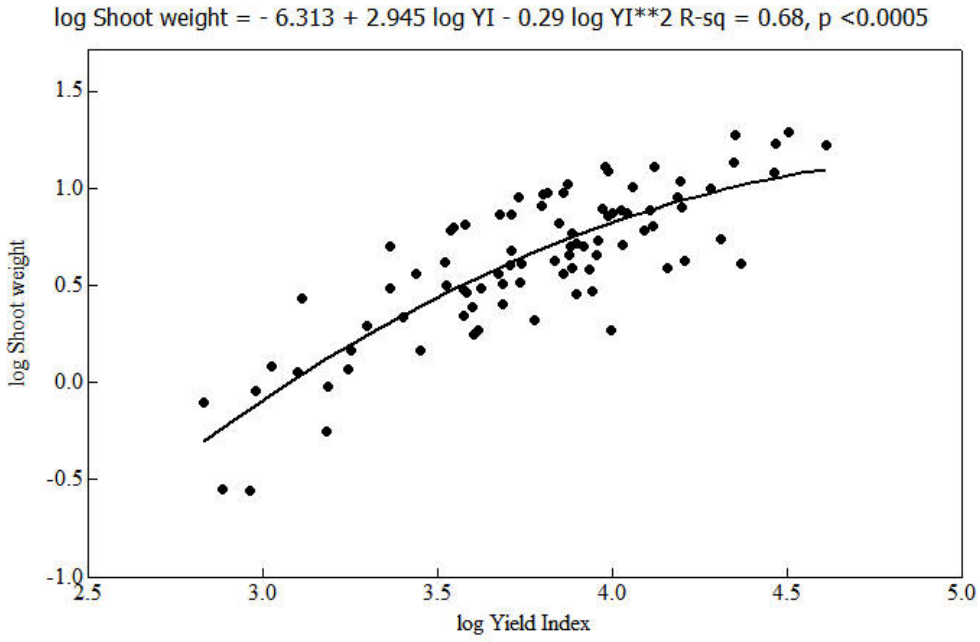


Figure 5.7: Regression of log YI against log shoot weight of seedlings and tillers harvested 6 months after the start of the experiment for a. *Allocasuarina littoralis*; b. *Eucalyptus obliqua* and; c *Poa labillardierei*.

Target plant Yield Index response

i. *Allocasuarina littoralis*

The Yield Index (YI) of *A. littoralis* seedlings when grown without neighbour plants had an average relative increase ($RGR_{(YI)}$) of 11.93 ± 1.95, as shown by the broken lines in Figure 5.8. However, unlike the effect on seedling height, no competition treatment was found to produce a greater $RGR_{(YI)}$ than that of the seedlings grown without neighbours. In particular, the neighbour treatment of 2A, 2P produced $RGRs_{(YI)}$ at both the 3.5 cm (2.6 ± 1.73) and 7.0 cm (2.2 ± 0.59) distance that were nearly five times less the $RGR_{(YI)}$ that occurred without neighbours.

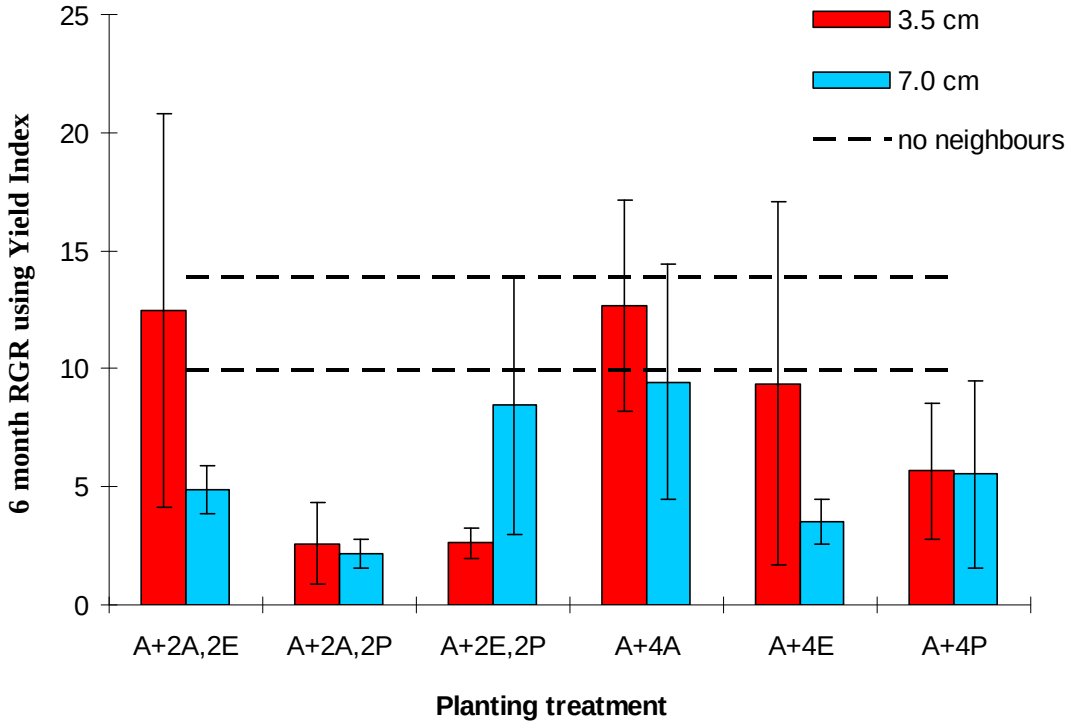


Figure 5.8: $RGR_{(YI)}$ of *Allocasuarina littoralis* planted with six combinations of neighbours at 2 distances. Columns show the mean $RGR_{(YI)}$ over 6 months ± standard error. The two broken lines indicate the range of the mean $RGR_{(YI)}$ ± one standard error when seedlings were grown without neighbours. A = *A. littoralis*, E = *E. obliqua*, P = *P. labillardierei*.

Due to non-normality of the data and the residuals, a GLM analysis could not be undertaken to test combined effects of treatment and distance on *A. littoralis* $RGR_{(YI)}$. However a non-parametric analysis found that there was a significant effect of treatment (Kruskal-Wallis test, $p = 0.032$, $H = 12.18$). Further analysis using the Mann-Whitney U test showed that target plant $RGR_{(YI)}$ was significantly higher when grown within an *Allocasuarina* monoculture than with three other neighbour combinations; 2A+2P ($p = 0.0018$), 2E+2P ($p = 0.037$), and 4E ($p = 0.0277$).

ii. *Eucalyptus obliqua*

Eucalyptus obliqua seedlings had a mean $RGR_{(YI)}$ of 4.53 ± 0.74 after 6 months of growth without neighbours (Figure 5.9). There was only one competition treatment, 4A at the 7.0 cm distance, which resulted in the target plant having a significantly higher $RGR_{(YI)}$ after 6 months than that of seedlings grown without neighbours, which corresponds to the effect of this treatment on seedling height (see Figure 5.5).

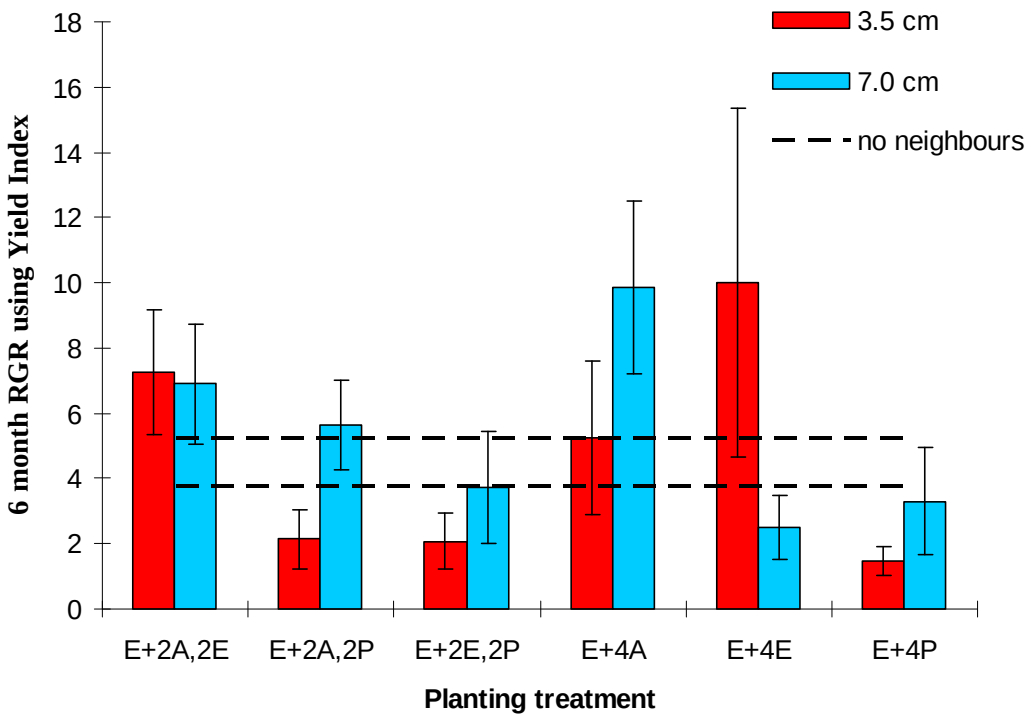


Figure 5.9: $RGR_{(YI)}$ of *Eucalyptus obliqua* planted with six combinations of neighbours at 2 distances. Columns show the mean $RGR_{(YI)}$ over 6 months ± standard error. The two broken lines indicate the range of the mean $RGR_{(YI)}$ ± one standard error when seedlings were grown without neighbours. A = *A. littoralis*, E = *E. obliqua*, P = *P. labillardierei*.

The interaction of treatment and distance was found not to be significant (GLM, F = 0.97, p = 0.448) on target plant $RGR_{(YI)}$ which differs from the effect it had on seedling height ($RGR_{(H)}$). However, the distance of neighbours (GLM, F = 4.46, p = 0.040) and neighbour combination (GLM, F = 3.57, p = 0.008) both had a significant impact on the $RGR_{(YI)}$ of target plants. Target plants had greater growth when neighbours were planted at a lower density and when grown with *A. littoralis*. The initial size of neighbours was also found to have a significant impact (GLM, F = 4.88, p = 0.032) on *E. obliqua* seedling $RGR_{(YI)}$, although this impact reduced over time to become insignificant after 6 months (GLM, F = 0.01, p = 0.941).

iii. *Poa labillardierei*

The mean $RGR_{(YI)}$ of *P. labillardierei* tillers after 6 months growth without neighbours was 25.52 ± 4.97, which was not significantly different to the $RGR_{(YI)}$ of *P. labillardierei* tillers grown with any type of neighbour combination (GLM, F = 1.32, p = 0.269). The interaction of treatment and distance was also insignificant (GLM, F = 1.36, p = 0.252), however the effect of distance alone was found to have an significant effect (GLM, F = 3.15, p = 0.081) on the $RGR_{(YI)}$ of target plants. This was not completely consistent with the significant effect that both distance and the interaction of treatment and distance had on the heights of *P. labillardierei* tillers (Figure 5.6).

As with *E. obliqua* seedlings, the initial size of neighbours was found to have a significant impact (GLM, F = 7.13, p = 0.010) on *P. labillardierei* $RGR_{(YI)}$, with the impact becoming insignificant after 6 months (GLM, F = 0.15, p = 0.695).

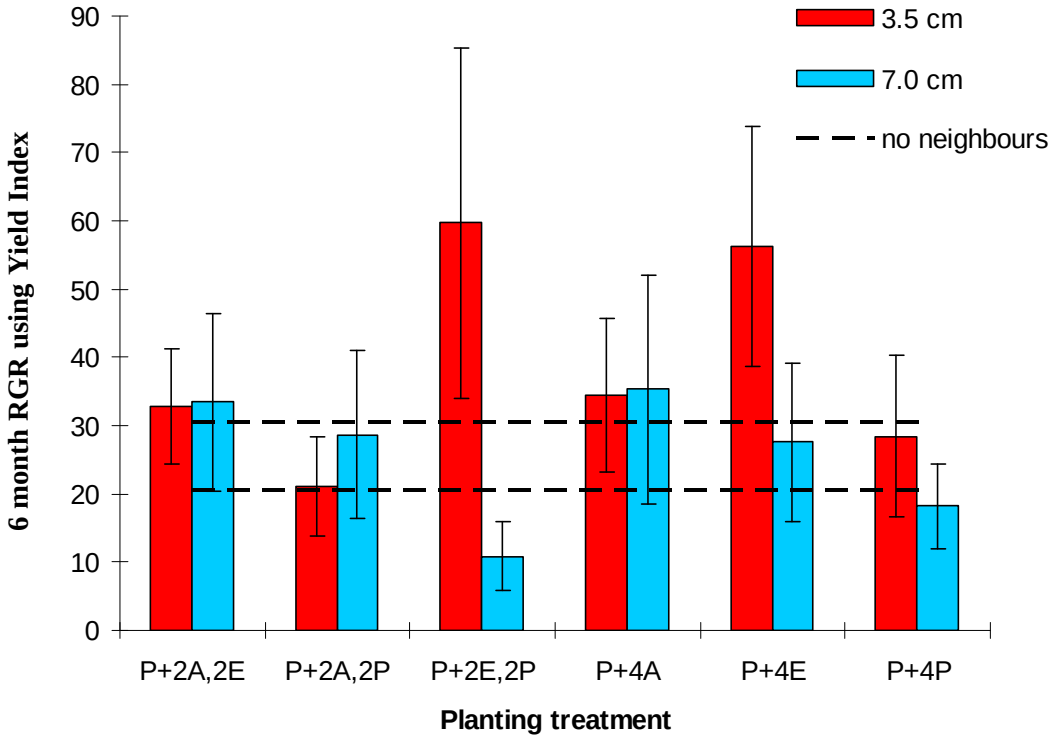


Figure 5.10: $RGR_{(YI)}$ of *Poa labillardierei* planted with six combinations of neighbours at 2 distances. Columns show the mean $RGR_{(YI)}$ over 6 months ± standard error. The two broken lines indicate the range of the mean $RGR_{(YI)}$ ± one standard error when seedlings were grown without neighbours. A = *A. littoralis*, E = *E. obliqua*, P = *P. labillardierei*.

Response of neighbours

The competitive response of the neighbour combinations (Figure 5.11) was found not to be significantly affected by the identity of the target species or the interaction of target plant identity and its distance, though distance of the target plant alone was significant (Table 5.2). All 6 combinations of neighbours had a greater increase in YI when the target plant was 3.5 cm away than when it was planted at a distance of 7.0 cm (Figure 5.12).

Table 5.2: Summary of the GLM analysis to look at the effects of distance and target species on the combined $RGR_{(YI)}$ of the four neighbours in each treatment.

Variable	DF	F	p-value
Distance	1	5.98	0.015
Target Species	2	1.64	0.197
Distance × Target species	2	0.18	0.834

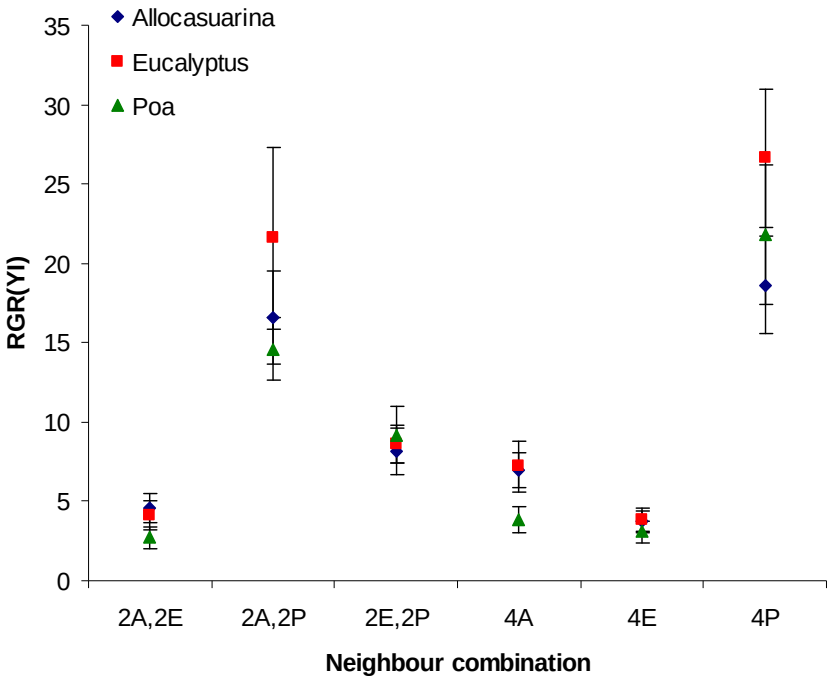


Figure 5.11: The effect of target plant identity on the combined $RGR_{(YI)}$ of four neighbour plants within each treatment. A = *A. littoralis*, E = *E. obliqua*, P = *P. labillardierei*.

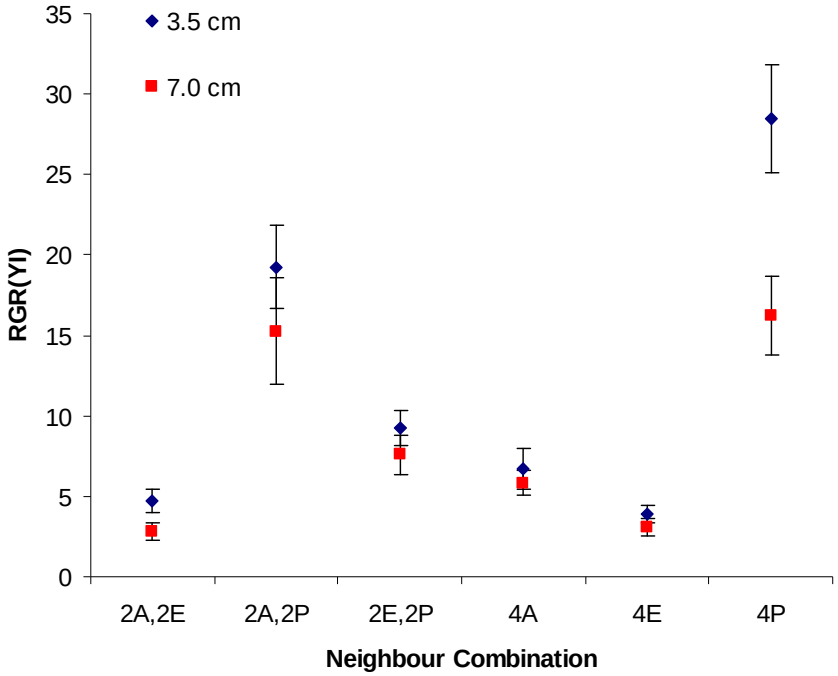


Figure 5.12: The combined $RGR_{(YI)}$ of four neighbour plants in each treatment when the target plant was 3.5 cm and 7.0 cm away. A = *A. littoralis*, E = *E. obliqua*, P = *P. labillardierei*.

Discussion

Effect of neighbour identity

The planting of *Allocasuarina littoralis* in monoculture at both densities produced the only significant height increase of *A. littoralis* target plants compared to that of seedlings grown in other treatment combinations. However, this positive increase was not reproduced when using Yield Index as a measure of plant performance. This variation in results could reflect that within monoculture stands, *A. littoralis* is responding to the competitive interactions of its neighbours by allocating more resources to stem height than leaf area. However, the results using Yield Index show that planting in monoculture did not have a significantly adverse effect on the above-ground biomass of *A. littoralis* in comparison to planting without neighbours, so any extra allocation of resources to height was not compensated by reduced biomass.

Eucalyptus obliqua also had significant increases in target plant height when planted with four *A. littoralis* seedlings as neighbours at both distances. However *E. obliqua* target plants also had a significant increase in Yield Index when it had *A. littoralis* seedlings as neighbours at the distance of 7.0 cm. The ability of trees and shrubs in the Casuarinaceae family to fix nitrogen and facilitate the growth of non-nitrogen fixing neighbours has been well documented (Hunter and Aarssen 1988; Binkley 1992; Maron and Connors 1996; DeBell *et al.* 1997; Barritt and Facelli 2001). This increase in growth of mixed stands of nitrogen fixing with non-nitrogen fixing plants has been recorded to occur on nitrogen deficient sites (DeBell *et al.* 1997) whereas on sites with ample supplies of nitrogen, mixed stands typically performed more poorly than pure stands of non-nitrogen fixing species (Binkley 1992).

Native heath and woodland species from Hawkesbury sandstone communities were found to increase in biomass (Thomson and Leishman 2004) after nutrient addition but this was accompanied by an overall reduction in species richness and density. In general, Thomson and Leishman (2004) found that the survival of native plants decreased with increasing concentrations of nitrogen and phosphorus. Forrester *et al.*

(2005) found that mixed plantations of *Eucalyptus globulus* with the nitrogen fixing tree species *Acacia mearnsii* produced significantly higher quantities of above ground biomass than monocultures, however, at high levels of nitrogen fertiliser *E. globulus* suppressed *A. mearnsii*.

In the present study it was expected that the percentage of total nitrogen would not be a limiting factor in seedling growth as the vegetation of the experimental location was dominated by exotic grasses with little to no regeneration of native plant species. However, the average percentage of total soil nitrogen recorded within the experimental plots was 0.15% ± 0.01, which is similar to that of the nitrogen levels of 0.13% found in native *E. obliqua* dry forest (Grant *et al.* 1995). Thus the facilitation of *A. littoralis* neighbours on the growth of target species at the study site is consistent with previous findings, that on sites of limited nitrogen soil concentrations, mixtures of nitrogen fixing plants will have a positive effect on the above ground growth of non-nitrogen fixing plants.

Eucalyptus target plant growth was also positively affected when grown in monoculture as well as mixtures with *A. littoralis* and itself. When neighbours were an initial distance of 3.5 cm, the combination of two *A. littoralis* plus two *E. obliqua* seedlings had a positive effect on *E. obliqua* target seedling height but not on Yield Index. This increase in height, without an increase in Yield Index, was also repeated when *E. obliqua* was planted in monoculture with its neighbours an initial distance of 3.5 cm. The allocation of resources to stem height was possibly a response of *E. obliqua* seedlings to competition for below ground resources, as competition for light is unlikely between even-aged monocultures. In their study on the effects of *E. obliqua* density on the growth of *Pinus radiata*, Bi and Turvey (1994a) found that the reduction in leaf area index to stem volume ratio of *P. radiata* was most likely from below ground resource depletion, caused by the increasing density of *E. obliqua*, as the reduction in leaf area index of *P. radiata* was not completely compensated by the increase in leaf area index of *E. obliqua*. Harper (1977) stated that the growth rate of a young plant is greatest if it establishes before its neighbours, enabling it to pre-empt resources. The increase in

height of *E. obliqua* without the corresponding increase in yield may also be the response of the target plant to pre-empt competition for light by allocating its resources to stem growth.

Neighbour identity alone did not have an impact on the height of *P. labillardierei* but the combination of neighbour type with distance was found to be significant. The greatest increase in *P. labillardierei* tiller height occurred when it had neighbour combinations of itself with either *E. obliqua* or *A. littoralis* at the initial distance of 3.5 cm. However, there was not any significant difference in the $RGR_{(YI)}$ of *Poa* when planted without neighbours in comparison to when it was planted with any type of neighbour combination at either distance. *P. labillardierei* has been shown to increase in growth in response to an increase in nitrogen and phosphorus (Groves *et al*. 1973) which could explain the increase in height that was recorded when it was planted with a nitrogen fixing neighbour such as *A. littoralis*. However, Groves *et al.* (2003) found that when planted in mixtures with exotic grasses the native perennial was unable to compete at low or high nitrogen levels. In terms of using plantings *of P. labillardierei* with neighbours to aid its competitiveness against exotic grasses, *A. littoralis* may be seen not to help *P. labillardierei*'s competitive ability through increasing nitrogen levels of the site. However, the increase in height *P. labillardierei* experienced when planted with *A. littoralis* may give it an advantage over exotic grasses in competing for light resources. The increase in height but not yield that *P. labillardierei* target plants experienced with a neighbour mixture of itself and *E. obliqua* at high density may also be explained by the target plant putting resources into height so as to out complete neighbours for light resources (Harper 1977).

Effect of neighbour size

The combination of two *A. littoralis* seedlings and two *P. labillardierei* tillers at both distances were shown to have a negative impact on A. *littoralis* target plant height and yield. *P. labillardierei* had the greatest growth rates during the experimental timeframe and thus may be able to suppress the growth of the smaller sized *A. littoralis* through its greater growth capacity. Neighbour combinations including *E. obliqua* seedlings also had an adverse effect on *A. littoralis* target height. *E. obliqua* seedlings were initially taller and had a greater biomass than *A. littoralis*.

Significant relationships between neighbour biomass and target plant performance have been observed to occur (Harper 1977; Goldberg and Landa 1991; Weigelt *et al.* 2002), showing a positive correlation of competition and biomass (Goldberg and Landa 1991). On the other hand there is also evidence that competitive interaction does not significantly depend on neighbouring plant biomass (Weigelt *et al.* 2002). Peltzer and Kochy (2001) found that light availability decreased linearly with increases in grass biomass, but soil resources did not vary with neighbour biomass and that overall the presence of neighbour species was more important in reducing resource levels than was neighbour biomass.

Interestingly, when planted with four *P. labillardierei* neighbours, *A. littoralis* target plant height was not adversely affected. This, along with the insignificant relationships found between *A. littoralis* target plant $RGR_{(H)}$ and the initial average height of its neighbours, as well as the average neighbour height at 6 months, seem to indicate that the competitive interaction between *A. littoralis* and its neighbours did not depend on neighbour size. This reflects findings by Kochy and Wilson (2000) that competitive interactions between seedlings of different growth forms are symmetric during the early stages of growth when seedlings are competing for soil resources rather than light resources.

As found with *A. littoralis,* target plant height of *E. obliqua* seedlings and *P. labillardierei* tillers were not correlated to the height of neighbours at the time of planting or after 6 months of growth. However, the yield of both species had a significant relationship with the initial biomass of neighbours. McConnaughay and Bazzaz (1990) also found that the height of the target individual was only moderately influenced by the identity of its neighbour, but in contrast to height results, they found that the accumulation of dry mass of the target individual was influenced by the neighbour treatment.

Despite this initial relationship with neighbour biomass, the effect reduced over time to become insignificant at 6 months for both *E. obliqua* seedlings and *P. labillardierei* tillers. This reflects the observation of Harper (1977) that plants at a higher density will experience stress from their neighbours early in their development. Whereas, a study on the effect of neighbours on *Poa pratensis* demonstrated that there appeared to be no general relationship between competition intensity and neighbour biomass (Reader *et al.* 1994).

The results observed in the current study indicate that interactions between the target plant and its neighbour's size varied over the length of the experiment. Interactions between plants are known to change with variations in abiotic conditions and changes in seedling physiology (Callaway 1998). As the experimental conditions changed from mid-summer to mid-winter, the combination of competitive interactions also likely changed, however by the end of the study, the size of the neighbour was observed to be less influential than the type of neighbour on target plant growth.

Effect of neighbour distance

The findings that *A. littoralis* height and biomass growth was not significantly affected by distance of neighbour indicate that neighbours at both distances were interacting with the target plant performance. In contrast, distance was found to change the competitive effects of neighbours on *E. obliqua* target plant height. When planted in monoculture with neighbours at a distance of 3.5 cm, *E. obliqua* had a significantly greater height than when planted without neighbours. However, when planted in monoculture with seedlings at a distance of 7.0 cm, target plant height was less than that of the control.

The yield of *E. obliqua* target plants was also significantly influenced by the distance of neighbours with target growth on average greater when neighbours were planted at the greater distance. This seems to indicate that having neighbours at a distance of 7.0 cm was more facilitative to *E. obliqua* growth, however this doesn't explain why when planted in a monoculture *E. obliqua* yield was five times greater at the 3.5 cm distance than when neighbours were planted at 7.0 cm. Although the combination of treatment and distance was not statistically significant for *E. obliqua* yield, the results indicate that the effect of distance was not consistent across the different neighbour combinations and that distance strongly influenced the effect of neighbour type.

A significant effect of distance, as well as distance in combination with neighbour type, was also observed for *P. labillardierei* height and yield, as closer neighbours resulted in taller *P. labillardierei* tillers. Although collectively *P. labillardierei* height was greater when neighbours were at a distance of 3.5 cm, neighbour combinations at 7.0 cm also increased height and yield, indicating, as with the *E. obliqua* target plants, neighbours at both distances were affecting growth. This combined effect of distance and neighbour type was most pronounced for the neighbour mixture containing *E. obliqua* and *P. labillardierei,* which only facilitated growth when planted at high density.

Conclusions

Although beneficial interactions among plants were observed for all target species when using height as a performance indicator, only *E. obliqua* had neighbours that positively affected its Yield Index. Having four *A. littoralis* seedlings as neighbours at the 3.5 cm distance had a positive effect on *E. obliqua* height, whereas *A. littoralis* neighbours at the distance of 7.0 cm had a positive affect on height and yield. This indicates that the facilitative effects of *A. littoralis* through increasing available soil nitrogen at the wider distance (or lower density) was not countered by the negative effect of competition for space. In turn, *A. littoralis* target plants were not adversely affected by *E. obliqua* neighbours, indicating that, under the current site conditions seedlings of the two species can be re-introduced simultaneously to encourage native plant re-establishment.

Co-existence between species has been found to occur with little competition between the benefactor and beneficiary (where a plant is positively affected by the conditions created by another plant) when understorey plants are planted with canopy trees (Hunter and Aarssen 1988). *E. obliqua* seedlings had an original height and size advantage over *A. littoralis* seedlings so along with the facilitative improvement of soil nitrogen availability by *A. littoralis* seedlings, complementary resource use may occur between the species because of a stratified canopy (Forrester *et al*. 2005; Kelty 2006). However, when the initial density was higher, *A. littoralis* neighbours did not facilitate an increase in *E. obliqua* yield, indicating that the facilitative effects of *A. littoralis* did not occur at neighbour densities where neighbour biomass may have reduced other resource levels such as soil water and light availability. On the other hand, as *A. littoralis* seedlings were shorter than *E. obliqua,* they were not likely to have reduced light availability in this case. Also it should be noted that *E. obliqua* yield was not negatively affected by *A. littoralis* seedlings at the close distance and so a complementary use of resources may be occurring at the higher density. Weigelt *et al*. (2002) found that only minimal changes of target biomass occur in competitive interactions between adult plants even if neighbour biomass changes substantially. Thus, once *E. obliqua* seedlings are established, their growth being facilitated by the nitrogen fixing abilities of *A. littoralis*,

increasing neighbour density is unlikely to negatively affect target plant growth. The improved *E. obliqua* performance with neighbours at the initial distance of 3.5 cm indicates that selecting an appropriate initial neighbour density can add to the facilitative effects of the neighbour type.

E. obliqua may also facilitate growth of *P. labillardierei* through the complementary use of resources, as planting *P. labillardierei* tillers with *E. obliqua* seedlings significantly improved the target plant height. In addition, target plant yield was also greater when the *E. obliqua* seedlings were planted at the close distance. As a neighbour, *E. obliqua* may alter the physical environment through limiting light which *P. labillardierei*, as a ground-cover species, may prefer over open conditions. However, this complementary use of resources was not evident when *E. obliqua* was the target plant and *P. labillardierei* tillers its neighbours. Although a groundcover species, *P. labillardierei* tillers were larger than seedlings of the shrub *A. littoralis* and adversely affected both *A. littoralis* and *E. obliqua* target plant yield, though the results were not consistent over both densities.

In the three species mixtures at the lower density, both *E. obliqua* and *A. littoralis* target plant growth did not vary significantly from plant growth without neighbours. This indicates that the competitive effects of a third species in a mixture may be able to reduce the competitive suppression of a dominant species (Hunter and Aarssen 1988). However at the closer distance, mixtures of *P. labillardierei* still inhibited plant growth. Kelty (2006) suggested that for highly diverse forest types sequential planting stages are necessary. Thus, to successfully re-introduce tree, shrub and groundcover mixtures under the current site conditions, it would be beneficial to establish *E. obliqua* and *A. littoralis* seedlings in two species mixtures of seedlings of the same age then re-introduce *P. labillardierei* tillers once *E. obliqua* and *A. littoralis* reach a canopy height that will not be over taken by *P. labillardierei*.

Chapter Six

Conclusions and implications of research

The results of the present study highlight that restoration of temperate sclerophyll vegetation on improved pastures requires overcoming the multiple factors that restrict native species establishment. The absence of natural recruitment, even with the aid of gaps created in the grass sward by disturbances, indicated that a reduced native seed supply in combination with the superior competitive abilities of exotic grasses were limiting factors. Furthermore, variation in the success of native species related to the variation in weed species, soil and drainage conditions within the one location, indicates the importance of site trials in determining the most successful strategies for restoration.

Under the conditions of the field trial, where soil moisture was not a limiting factor, direct seeding proved most successful in drier plots where weed control was prolonged by the combination of weed removal and grazing by native and introduced herbivores. However selective grazing on seedlings grown in tubestock was observed at the field site, so, although grazing of weeds facilitated native seedling emergence, protection from grazing may be needed 1 to 2 years after sowing when seedlings have reached a height above the grass sward. Direct seeding trials in central NSW (Geeves *et al.* 2008) also observed that macropod damage appeared to have its major effect on the growth of seedlings rather than survival. The broader implication of these findings is that successful direct seeding on sites where weed competition is a limiting factor on native emergence can be achieved when pre-sowing weed control is sustained by macropod grazing, although, once seedlings have established, protection from grazing may be needed facilitate greater growth.

Poor emergence of *Eucalyptus obliqua* and *Allocasuarina* seedlings from seed indicated that, even with ground preparation, it is difficult to produce suitable microsite conditions within improved pasture for native species germination. The results from the direct

seeding trial indicate the importance of identifying the combination of factors limiting native reestablishment before implementing a restoration strategy over a wider area. At the current site implementing large scale direct seeding is likely to result in high ratios of *Acacia* to *Eucalyptus*, resulting in reconstructed vegetation which does not biologically or structurally represent the native forest being restored.

Where greater levels of intervention are needed, such as reintroducing native seedlings, this generally corresponds with a greater investment in resources. Therefore site specific information is necessary to determine how best to allocate resources. In the current study, ascertaining the native vegetation communities which were present prior to agricultural disturbance was of great importance for setting restoration goals for each of the dominant weed communities present at the site. The variety of floristic, soil and drainage conditions of the different weed communities indicated that successful restoration needed to use species whose natural distribution range matches the site conditions.

The results from the seedling reintroduction trial (Chapter 4) clearly demonstrated that allocating resources to growing seedlings from locally sourced seeds, and planting seedlings in sites that correspond to its ecological range, can result in successful seedling performance irrespective of the use of weed control. Thus, on a site where weed competition is a limiting factor on native recruitment, this competition can be overcome, without the cost of weed control, by selecting species that suit the site conditions. However, basing species selection on natural ecological ranges may not be successful on sites which have been greatly modified from pre-disturbance conditions, for example, where areas are affected by dryland salinity, and reconstruction may need to rely on using species with specific tolerances to the new conditions.

Where the task of restorationists is to initiate successional development by replanting with native species the restoration of understorey species is usually left to secondary succession. There is thus a need for experimental work to determine whether the recovery of sclerophyll forest on abandoned farmland can be accelerated by the

simultaneous re-introduction of tree, shrub and groundcover species. At the current restoration site it was found that *Eucalyptus obliqua* seedling performance can be facilitated when planted in mixtures with *Allocasuarina littoralis*. Furthermore, the results of the experiment implied that sequential planting of species mixtures may produce even more successful results through complementary use of resources. Due to the complexities of interactions involved in species mixtures more research is needed in different environments and with mixtures that contain a greater complement of species to predict the success of planting mixtures under different conditions. However, the present research does show that the simultaneous planting of species mixtures may be an efficient means of accelerating the recovery of sclerophyll forest biodiversity and structure through ameliorating soil conditions and depressing weed growth.

The present study has also demonstrated that, where the landscape has been modified (i.e. improved pasture), experiments can help determine how best to reintroduce the native species and that this work has the potential to inform restoration strategies that can consistently work over a wider geographic range.

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